REPORTS OF THE OAK RIDGE DOSE RECONSTRUCTION, Vol. 2
The Report of Project Task 2 • July 1999

Mercury Releases from Lithium Enrichment at the Oak Ridge Y-12 Plant – a Reconstruction of Historical Releases and Off-Site Doses and Health Risks

Submitted to the Tennessee Department of Health by

ChemRisk
A Servicemaster McLaren/Hart

OAK RIDGE HEALTH STUDIES
OAK RIDGE DOSE RECONSTRUCTION

– TASK 2 REPORT –

MERCURY RELEASES FROM LITHIUM ENRICHMENT AT THE OAK RIDGE Y-12 PLANT– A RECONSTRUCTION OF HISTORICAL RELEASES AND OFF-SITE DOSES AND HEALTH RISKS

July 1999

Submitted to the Tennessee Department of Health by

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

Project Manager

Thomas E. Widner[1]

Task 2 Manager

Thomas R. Mongan[1]

Principal Authors

Gretchen M. Bruce[2]
Susan M. Flack[3]
Thomas R. Mongan
Thomas E. Widner

---

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

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

[3] ChemRisk®, a Service of McLaren/Hart Environmental Services, Inc., 1544 North Street, Boulder, Colorado 80304.

TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

TABLE OF CONTENTS (Continued)

TABLE OF CONTENTS (Continued)

LIST OF APPENDICES (Volume 2A)

LIST OF FIGURES

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF FIGURES (Continued)

LIST OF TABLES

LIST OF TABLES (Continued)

LIST OF TABLES (Continued)

LIST OF TABLES (Continued)

GLOSSARY

ADP - the Alloy Development Project (or Plant) at the Y-12 Plant.

alloy - a former Y-12 codeword for lithium; also, a substance composed of two or more metals, or metal(s) and a nonmetal, that are intimately mixed.

amalgam - an alloy of mercury with another metal.

angler - usually a person who fishes with line and hook. Used in this report to mean any fish consumer, including members of a family who ate fish caught by another person.

aspen - a former Y-12 codeword for the ^{6}Li isotope of lithium.

bgs - an abbreviation of "below ground surface".

bioaccessibility- the fraction of a chemical substance that desorbs from its matrix under physiological conditions and is available for absorption into the bloodstream

bioavailability - the fraction of a chemical substance that is absorbed into the bloodstream and available to cause toxicity; may be described as the product of the bioaccessibility and absorption of the chemical.

chloralkali plants - industrial plants that can use an electrochemical process including a mercury cathode to produce chlorine gas from sodium brine; currently, the largest user of mercury in the U.S. (typically 3 tons of mercury per cell, 100 cells per plant).

CNS- the central nervous system (brain and spinal cord)

Colex - a column chemical exchange process for enrichment of lithium in its ^{6}Li component.

composite sample - a combination of samples taken over a set period of time (or area/ depth) that are analyzed as one sample.

deposition velocity - the ratio of the flux of material to the surface (mg m^{-2} s^{-1}) to the air concentration (mg m^{-3}).

DGM - "dissolved gaseous mercury"; mercury in streams and lakes that exists partially in a volatile phase.

DOE - U.S. Department of Energy.

dose - the amount of a substance taken in by an individual over a period of time from a variety of sources, including food, water, soil, and air, by such exposure pathways as ingestion, inhalation, or absorption through the skin. In this assessment, doses are described as daily intake rates averaged over periods of one year, and presented on a per kilogram of body weight basis.

EFPC - East Fork Poplar Creek.

elemental mercury - a shiny, silver-white, extremely dense, odorless liquid, that is the familiar species of mercury found in thermometers; tends to be relatively insoluble in water. Symbolized by the notation Hg^{0}.

Elex - an electrical exchange process for enrichment of lithium in its ^{6}Li component.

GLOSSARY

equilibrium- a state of balance between opposing forces; in chemistry, the stage in a reversible chemical reaction at which the products of the forward reaction are consumed by the reverse reaction at the same rate as they are formed.

evasion - the escaping of a vapor from a liquid.

exposure routes - mechanisms or pathways through which an individual may contact contaminants in environmental media (*e.g.*, air, soil, or water). Some commonly encountered exposure routes are: inhalation of contaminated air, ingestion of contaminated soil, water, and food stuffs, and dermal contact with contaminated soil or water.

foliar uptake - uptake of a substance through the leaves of a plant.

gavage - forced feeding, as by a flexible tube and pump, often used in animal toxicity studies.

health effect endpoint - a defined measure of an adverse health effect, such as cancer, elevated levels of substances in blood or urine, or tremors.

Hermex - a process for purifying uranium by dissolving it in boiling mercury and recovering it as uranium mercuride. Tested at the X-10 site in the 1950s.

inorganic mercury - a group of compounds or "salts" present after the mercury ion (Hg^{+} or Hg^{2+}) forms a chemical bond with elements other than carbon, such as chlorine or sulfur, or with hydroxide (OH^{-}) ions. Elemental mercury and inorganic mercury compounds are often grouped under the generic term "inorganic mercury"; however, in this report, inorganic mercury refers only to mercury salts.

in utero- in the uterus, before birth

in vitro- outside the living body of an animal or plant, and in an artificial environment

in vivo- in the living body of an animal or plant

ISCST3 - Industrial Source Complex Short Term (version 3); a USEPA-approved air dispersion model.

isotopes - forms of a chemical element having the same number of protons, but different numbers of neutrons, and therefore different atomic weights (for example, ^{6}Li and ^{7}Li).

Large-Scale Review - a Y-12 classified document review program conducted in 1994 that supported U.S. DOE efforts to provide environmental, safety, and health information to the general public, comply with the USDOE Tennessee Oversight Agreement, and declassify or downgrade document holdings.

lithium deuteride - a light element fuel used in thermonuclear weapons. Deuterium is an isotope of hydrogen. Symbolized by the notation "LiD".

LOAEL - "lowest observed adverse effect level"; the lowest dose of a substance used in a toxicity study that produced statistically or biologically significant differences between the frequencies or severity of adverse effects observed in exposed and control populations of test animals or humans.

marble - a former Y-12 codeword for the ^{7}Li isotope of lithium.

GLOSSARY

Mercury Files - the collection of documents assembled by the Mercury Task Force in 1983.

Mercury Task Force - a group appointed by the Y-12 Plant Manager in 1983 to collect historical data on mercury accountability, study mercury salvage and recovery, and summarize studies of mercury impacts on worker health and the environment.

Metallex - a process for purifying thorium and uranium metal from their compounds using sodium amalgam. Tested at the X-10 site in the 1950s.

metallic mercury - an alternate name for elemental mercury.

methylmercury - an organic mercury compound, produced by bacteria and chemical processes, that is easily absorbed by fish and other aquatic fauna. Can bioaccumulate to higher concentrations than in the surrounding media.

Monte Carlo Simulation - a mathematical technique that uses random selection to simulate the effect of uncertain knowledge of input parameters on the answer provided by an equation or model.

MRL - the Agency for Toxic Substances and Disease Registry's "minimal risk level"; an estimate of daily human exposure to a dose of a chemical that is likely to be without an appreciable risk of adverse noncancerous effects over a specified period of duration; intended to acquaint health professionals with exposure levels at which adverse health effects are not expected to occur in humans, not intended to support regulatory action.

NOAEL - "no observed adverse effect level"; the highest dose of a substance used in a toxicity study that produced no statistically or biologically significant differences between the frequencies or severity of adverse effects observed in exposed and control populations of test animals or humans.

northing - a distance toward the north from a specified point.

Orex - an organic exchange process for enrichment of lithium in its ^{6}Li component.

organic mercury - a group of compounds present after mercury combines in a chemical bond with carbon. An example is methylmercury.

ORO - the U.S. Department of Energy's Oak Ridge Operations.

ORR - the Oak Ridge Reservation.

oxidation- the process of removing one or more electrons from an atom, ion, or molecule, as when Hg^{0} is oxidized to Hg^{2+} by the removal of two (negatively charged) electrons.

paresthesia - a tingling sensation in the extremities; one of the symptoms of mercury toxicity.

PDF - "probability density function"; a subjectively defined function that quantitavely expresses the state of knowledge about a parameter value by characterizing the degree of belief that the true but unknown value of the parameter lies within a specified range of values for that parameter.

percentiles - if a large set of data is arranged from its smallest value to its largest, and this list is divided into 100 classes containing nearly equal numbers of data points, then each percentile represents the highest value within that class. Thus 5% of the data are less than or equal to the 5th percentile, and approximately 95% of the data are greater than or equal to the 5th percentile. The median is defined as the 50th percentile, which divides the data (approximately) into halves.

receptor location - a geographic location of individuals within the assessment domain where exposure concentrations are estimated.

reduction- the process of adding one or more electrons to an atom, ion, or molecule, as when Hg^{2+} is reduced to Hg^{0} by the addition of two (negatively charged) electrons.

relative bioavailability - the ratio between the bioavailability of a substance in a person exposed in the environment and the bioavailability of that substance administered to an animal in a toxicity study (due to differences in exposure conditions between a laboratory study and real world exposure).

Reservation - in this report, refers to the Oak Ridge Reservation.

RfD - the U.S. Environmental Protection Agency's "reference dose"; a dose rate of a chemical that is not expected to cause adverse health effects over a lifetime of daily exposure in humans (including sensitive subgroups). Expressed in units of milligrams per kilogram of body weight per day; mg kg^{-1} day^{-1}.

risk - the probability of a deleterious health effect, such as cancer, being induced.

solvent - a former Y-12 codeword for mercury.

source term - refers to the quantity, chemical and physical form, and time history of a contaminant released to the environment from a facility.

southing - a distance toward the south from a specified point.

terrestrial biota - animals or plants that live on land, such as cattle.

transect - a sample area, usually a long strip, that cuts across a larger area. For example, in the East Fork Poplar Creek floodplain, soil samples were collected from floodplain transects perpendicular to the creek.

uncertainty - a lack of knowledge or certainty about the true but unknown value of a parameter. Can be expressed using a quantitative probability density function (PDF). Uncertainties in reconstructing doses can arise from a number of sources, including uncertainties about the accuracy of historical measurements, absence of data at exposures points, lack of knowledge about some physical processes and operational procedures, and the approximate nature of mathematical models used to predict the transport of released materials.

weir - a small dam in a river or stream where flow rate can be estimated by measuring the height of flowing water.

EXECUTIVE SUMMARY

Between 1950 and 1963, while lithium was being enriched in its lithium-6 component for use in thermonuclear weapons, many tons of mercury were released to the air and surface waters from the Y-12 Plant on the Oak Ridge Reservation in Oak Ridge, Tennessee. Preliminary investigations in the Oak Ridge Dose Reconstruction Feasibility Study indicated that mercury releases from operations at the Y-12 Plant likely resulted in the highest potential non-cancer health risks associated with historical activities on the Oak Ridge Reservation. Because of that finding, Task 2 of the Oak Ridge Dose Reconstruction was initiated by the Tennessee Department of Health and the Oak Ridge Health Agreement Steering Panel to bring about a detailed, independent[1] investigation of potential off-site doses and health risks from historical releases of mercury from the Y-12 Plant. This report documents the methods and results of that detailed investigation, which had the following objectives:

- to describe (and independently quantify) past releases of mercury from the Reservation;
- to characterize historical environmental concentrations of mercury from those releases;
- to define potential pathways of human exposure to mercury that have been in place;
- to describe populations that were potentially exposed;
- to estimate human exposures and doses that were potentially received; and
- to estimate human health hazards, to put the historical dose estimates in perspective.

The Oak Ridge processes that used the most mercury were the lithium separation operations conducted at the Y-12 Plant in the 1950s and 1960s. Lithium separation operations included three production facilities, requiring over 24 million pounds of mercury. While Colex process facilities (which used a column-based exchange process) at Y-12 were most significant in terms of mercury releases, the project team also documented the historical presence of much lower quantities of mercury in facilities built to test or demonstrate other processes for lithium enrichment, in instrumentation across the Reservation, in some nuclear weapon components, in processes for chemical recovery or decontamination of nuclear materials, and in the coal burned in on-site steam plants.

The project team's review of lithium enrichment operations and mercury releases began with examination of records assembled by members of a 1983 task force appointed to investigate and quantify mercury releases from Y-12. The project team's information gathering activities centered on interviews with

[1] The project was not under the technical direction of the U.S. Department of Energy, but was funded by a U.S. DOE grant to the Tennessee Department of Health.

members of the 1983 Mercury Task Force, review of classified and unclassified versions of the Task Force's report, and review of thousands of Task Force files and documents archived in the Y-12 Records Center. While the Task 2 information gathering process began with review of 1983 Mercury Task Force information, the Task 2 investigation differed from the 1983 Mercury Task Force study in that:

- The Task 2 team conducted a more thorough records review, including an extensive search of boxes of inactive Health and Safety records.

- The Task 2 team took additional steps to verify the data used to estimate historical mercury releases. For example, actual building ventilation rates were estimated based on review of historical drawings and documents.

- The Task 2 team identified and examined in detail the bases of the 1983 Mercury Task Force's release estimates, and revised the release estimates in a number of areas where the Task 2 team was able to assemble more complete information.

- The Task 2 team identified the references that support the 1983 Mercury Task Force's release estimates and had them made available to the public. These same documents form the basis of the Task 2 team's release estimates.

- The Task 2 team did not use estimates of unaccounted for mercury inventory in their estimates of offsite releases and doses. The Task 2 team estimated 62,000 pounds more mercury released to air and water than estimated by the 1983 Mercury Task Force. However, this increase represents only 3% of the unaccounted for mercury inventory.

Releases of airborne mercury from the Y-12 Plant were primarily a result of building ventilation systems installed to lower the concentration of mercury vapor inhaled by workers in the lithium enrichment facilities. While airborne mercury in the Y-12 Plant's exhausts was not routinely monitored, the Task 2 team located thousands of measurements of mercury in indoor air made between 1953 and 1962. The team obtained building ventilation rates from historical engineering drawings through the assistance of a former Y-12 Plant ventilation engineer, and used the estimated ventilation rates and measured concentrations of mercury in building air to estimate historical release rates. The Mercury Task Force used a similar method, but underestimated flow rates in a key Colex building by 50%. The Task 2 team estimated air emissions from four lithium enrichment buildings and a mercury recovery facility. Mercury emissions from the Y-12 steam plants were also estimated based on estimates of the natural mercury content of the coal and rates of coal consumption. Using corrected air concentrations and ventilation rates and including more mercury sources, the Task 2 team accounted for about 73,000 pounds of airborne mercury released**S** about 22,000 pounds more than the Mercury Task Force, or an increase of 43%. Figure ES-1 presents the Task 2 team's release estimates and those by the Mercury Task Force.

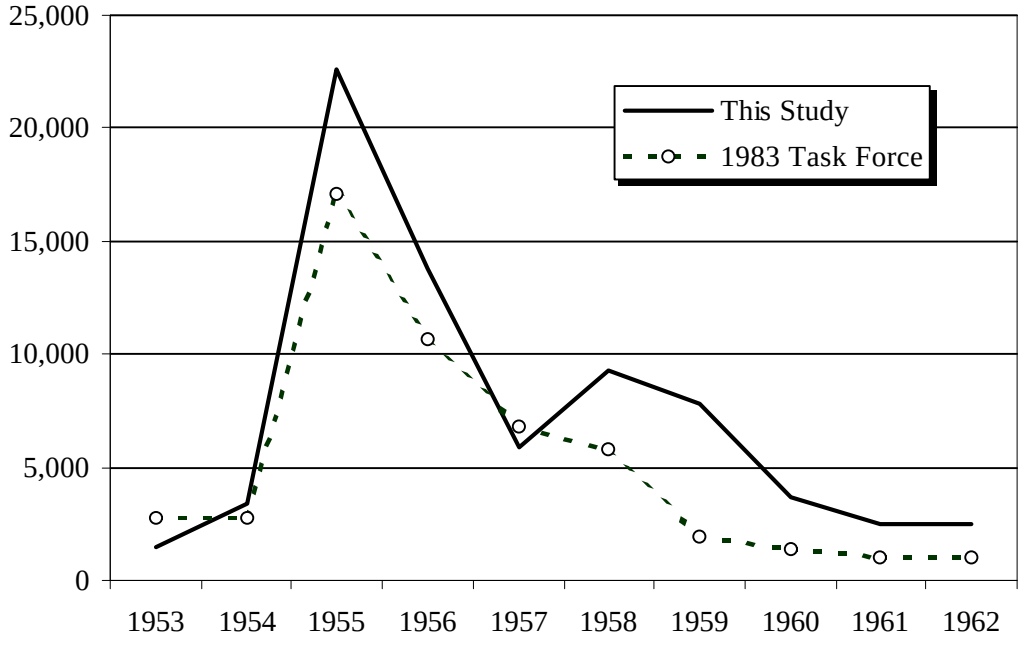


Figure ES-1: Annual Airborne Mercury Release Estimates by the Task 2 Project Team and the 1983 Mercury Task Force (pounds)

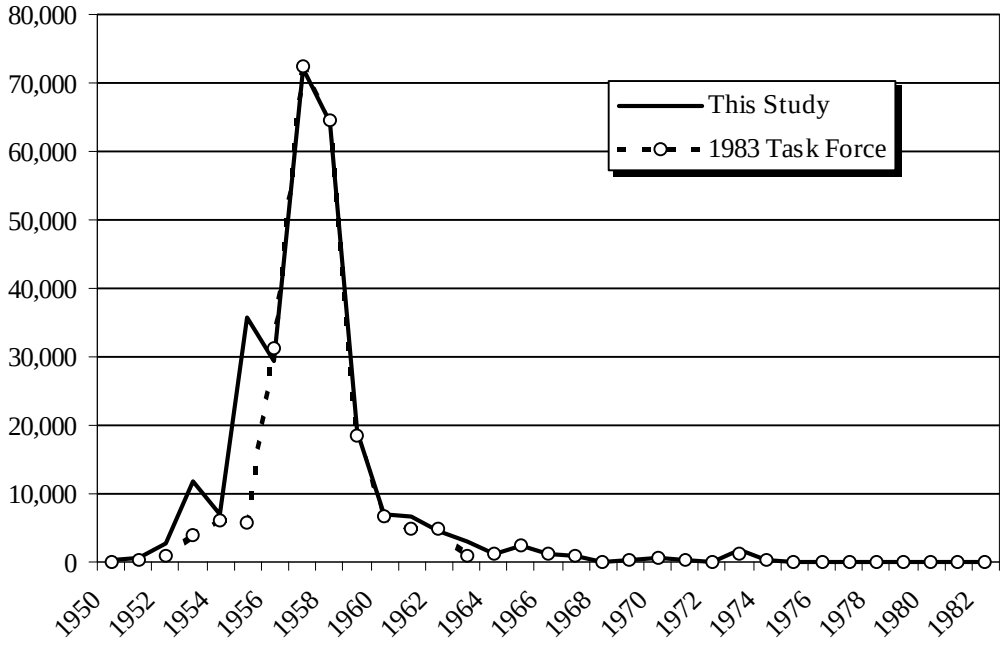


Figure ES-2: Annual Waterborne Mercury Release Estimates by the Task 2 Project Team and the 1983 Mercury Task Force (pounds)

Releases of waterborne mercury from the Y-12 Plant entered East Fork Poplar Creek (EFPC), and were largely a result of an early process in which mercury was washed with nitric acid. The waters of EFPC have been routinely sampled and analyzed for mercury since 1953, leading to what may be the longest record of mercury release from any site in the world. After finding measurement results for 1953 through 1993 from a number of sources, the Task 2 team compared the data, cross-checked values, and made corrections where mathematical errors had been made. Task 2 investigators also collected EFPC flow rate measurement data from numerous sources and were able to assemble a more complete data set than the 1983 Mercury Task Force used. With the more complete concentration and flow rate records, the Task 2 team accounted for about 280,000 pounds[1] of mercury released to EFPC from 1950 to 1993**S** about 44,000 more than officially reported. This 18% increase in the estimate of waterborne releases (the surface water "source term") is primarily due to reevaluation of releases between 1953 and 1955, as shown in Figure ES-2.

Mercury used in lithium isotope separations operations at Y-12 was elemental mercury, the familiar form of mercury commonly found in thermometers. When mercury is released to the environment from industrial processes, however, it can be converted in the environment to several different forms or *species*. The three primary forms of mercury that are found in the environment are elemental mercury (the dominant form in air, because of the tendency of elemental mercury to volatilize, and found also in soil, water, and food), inorganic mercury (or mercury "salts", found in soil, water, and food), and organic mercury (commonly found in fish as methylmercury). Each of these forms behaves differently in the environment and has been associated with different health effects in people and animals who were exposed to high concentrations. Because of differences in the behavior of these forms in the environment and differences in the potential adverse health effects following exposure, the Task 2 team evaluated each of the three forms of mercury separately.

In the calculations of estimated off-site doses from historical mercury releases, the Task 2 team selected a number of different geographic locations and types of potentially exposed people to investigate the ways in which doses likely varied as a function of location and of the characteristics and activities of the exposed individuals. These populations of interest include those likely to have received the highest exposures, due to their lifestyles and their proximity to release points and areas of high concentrations, and those with lower or more typical exposures reflective of larger portions of the general population. Because the rate of exposure to mercury and susceptibility to effects of exposure may vary as a function of age, exposures were evaluated for two age groups– adults and children. Exposures to methylmercury in fish were also evaluated for unborn children (*in utero* exposure) because toxicity studies have shown that unborn children may be particularly susceptible to adverse health effects when their mothers consume contaminated fish during pregnancy.

Because waterborne releases from Y-12 flowed into the waters of EFPC through residential and commercial sections of the city of Oak Ridge, the Task 2 team estimated doses and risks for the following populations:

[1] A volume of about 12 cubic yards (a cubic yard of mercury weighs approximately 22,800 pounds).

- *Oak Ridge Community Residents*– who lived near the EFPC floodplain and may have inhaled mercury volatilized from EFPC (assumed to be elemental mercury) and consumed homegrown fruits and vegetables contaminated by the airborne mercury volatilized from EFPC (assumed to convert within the plant to inorganic mercury). This group was assumed to have lived within one-half mile of EFPC in the western end of the city of Oak Ridge. Although exposures to individuals in this group likely varied somewhat depending on their location relative to the creek, the Task 2 team selected two discrete locations as representative of exposures to this group. The total size of this population between 1950 and 1990 was estimated to be between 15,000 and 30,000 individuals.

- *The Scarboro Community*– members of which inhaled mercury from the Y-12 Plant and EFPC (assumed to be elemental mercury), and could have traveled a short distance to EFPC for fishing (assumed to have been a route of exposure to methylmercury) or other recreational activities, and consumed homegrown fruits and vegetables contaminated by the airborne mercury (exposures through these pathways were assumed to be to inorganic mercury due to conversion of elemental mercury to inorganic in the plants). This has historically been the closest residential area to the Y-12 facility. The total size of this population between 1950 and 1990 was estimated to be between 6,000 and 10,000 individuals.

- *Robertsville School Children*– who attended a junior high school near the EFPC floodplain and inhaled mercury volatilized from EFPC (assumed to be elemental mercury) and came in contact with mercury in floodplain soil (assumed to be inorganic mercury). The total size of this population between 1950 and 1990 was estimated to be between 20,000 and 30,000 students. Based on interviews with local residents, some children in this age group were also assumed to recreate in EFPC for more significant periods of time, and come in contact with EFPC water and sediment (assumed to be inorganic mercury). The estimated size of this subpopulation between 1950 and 1990 was estimated to be between 100 and 300 children.

- *The EFPC Floodplain Farm Family*– who resided adjacent to the floodplain, farmed in or near the floodplain, grew fruits and vegetables (assumed to be contaminated by airborne elemental mercury and inorganic mercury from soil), raised beef and/or dairy cattle (assumed to take up mercury from air, EFPC water, and soil), and fished for recreation. This is not a hypothetical exposure scenario– land-use investigations indicate that several families resided adjacent to the floodplain and practiced these activities. The total size of this population between 1950 and 1990 was estimated to be between 40 and 200

Because meteorological studies indicate that the predominant direction of air flow from the Y-12 Plant is along Union Valley toward the northeast into Wolf Valley, the Task 2 team evaluated doses and risks for the following population:

- *Wolf Valley Residents***S** who lived in the area of the nearest dwelling historically located along the extension of Union Valley on the opposite side of the Clinch River, northeast of the Y-12 Plant. These individuals could have inhaled airborne mercury from Y-12 (assumed to be elemental mercury), grown fruits and vegetables (assumed to be contaminated by airborne mercury that was converted in the plant to inorganic mercury), and consumed milk and meat from "backyard" cattle that consumed pasture contaminated by airborne mercury. Exposures to this group were evaluated for 1953 to 1962, since this was the period of significant air releases from Y-12. The size of this population between 1953 and 1962 was estimated to be between 30 and 100 individuals.

Because of its close proximity to the Y-12 Plant, the Task 2 team also evaluated exposures of residents of the *Scarboro Community* to mercury in airborne releases from Y-12 between 1953 and 1962. While meteorological studies indicate that winds around the Y-12 Plant predominantly follow Bear Creek Valley and Union Valley, meteorological studies and ambient air monitoring programs (such as for uranium releases from the Y-12 Plant) indicate that the local ridges are not perfect barriers– it appears that some degree of transport of airborne effluents from the Y-12 Plant over Pine Ridge and into adjacent valleys does occur.

Because measurements since 1970 have shown that fish collected downstream of EFPC (that is, in Poplar Creek, the Clinch River, and Watts Bar Reservoir) contain elevated levels of mercury, the Task 2 team estimated doses to individuals who historically caught and consumed fish from these waterways. Exposures to mercury in fish, assumed to be methylmercury, were evaluated for three categories of fish consumers for Poplar Creek/ Clinch River and Watts Bar Reservoir, based on the number of fish meals consumed per year:

- *Category 1***S** Assumed to consume >1 to 2.5 fish meals per week (equivalent to about 24 to 61 g d^{-1} based on an average meal size of 170 g)

- *Category 2***S** Assumed to consume > 0.33 to 1 fish meal per week (or more than 1 meal every 3 weeks to 1 meal per week, equivalent to about 8 to 24 g d^{-1} based on an average meal size of 170 g)

- *Category 3***S** Assumed to consume 0.04 to 0.33 fish meals per week (or 1 meal every six months to 1 meal every 3 weeks, equivalent to about 0.97 to 8 g d^{-1} based on an average meal size of 170 g).

The exposure pathways evaluated for each population of interest and the species of mercury to which it is assumed exposure occurred through each pathway are summarized in Table ES-1.

Table ES-1: Exposure Pathways for Which Mercury Doses were Estimated for Each Population of Interest in the Task 2 Dose Reconstruction

Exposure Pathway	Mercury Species	Wolf Valley Resident	Scarboro Community Resident	Robertsville School- GeneralStudent	Robertsville School- Student Recreator	EFPC Floodplain Farm Family	Community Populations (2)	Clinch River/Poplar Creek Fish Consumer	Watts Bar Reservoir Fish Consumer
Air pathways									
Inhalation	*Elemental*	x [a]	x [b]	x [c]	x [c]	x [c]	x [c]		
Fruit/vegetable consumption	*Inorganic*	x [a]	x [b]			x [c]	x [c]		
Milk consumption	*Inorganic*	x [a]				x [c]			
Beef consumption	*Inorganic*	x [a]				x [c]			
Soil pathways									
Soil ingestion	*Inorganic*		x	x	x	x			
Skin contact with soil	*Inorganic*		x	x	x	x			
Vegetable consumption	*Inorganic*		x			x			
Milk consumption	*Inorganic*					x			
Beef consumption	*Inorganic*					x			
Sediment pathways									
Sediment ingestion	*Inorganic*		x		x	x			
Skin contact with sediment	*Inorganic*		x		x	x			
Surface water pathways									
Incidental ingestion of water	*Inorganic*		x		x	x			
Skin contact with water	*Inorganic*		x		x	x			
Milk consumption	*Inorganic*								
Beef consumption	*Inorganic*								
Fish consumption	*Methyl*		x			x		x	x

a Evaluated for direct airborne releases of mercury from Y-12

b For 1953-1962, evaluated for both direct airborne releases of mercury from Y-12 and volatilization of mercury from EFPC; for the remaining years, evaluated for volatilization of mercury from EFPC

c Evaluated for volatilization of mercury from EFPC

The Task 2 team based estimates of mercury exposures to each of these populations on historical and current measurements of mercury in different environmental media, or on data describing historical mercury releases from Y-12 and modeling of these releases to the locations of the exposure populations.

The Task 2 team estimated concentrations of mercury in the waters of EFPC at downstream locations based on independently verified measurements of concentrations and flow rates near Y-12, collected between 1950 and 1990, and application of factors to account for downstream reductions in concentrations due to mercury loss to other compartments (through adherence to sediment or volatilization to air) and dilution of water concentrations by additional inflow to the creek.

The Task 2 team calculated mercury concentrations in air at the *Wolf Valley Resident* location, due to direct releases from the Y-12 Plant, using independently verified estimates of annual releases to the air for 1953 to 1962. The Task 2 team used the Industrial Source Complex Short Term (ISCST3) air dispersion model and meteorological data collected on the Y-12 site to predict how these releases were carried off-site and to estimate annual average air concentrations at this location.

No measurements of mercury concentrations in air have been made in the Scarboro Community. However, measurements of airborne uranium, another contaminant historically released from Y-12 (and evaluated in Task 6 of the Oak Ridge Dose Reconstruction) were made at Scarboro in recent years (1986-1995), suggesting that some fraction of the airborne releases from Y-12 is transported over Pine Ridge. The Task 2 team estimated mercury concentrations in air at the Scarboro Community resulting from direct airborne releases from Y-12 by determining the ratio between measurements of airborne uranium in the Scarboro area and estimates of uranium releases from Y-12 developed by the Oak Ridge Dose Reconstruction Task 6 team. Empirical P/Qs (s m^{-3}) were estimated by dividing the Scarboro uranium air concentration (pCi m^{-3}) by the uranium release rate (pCi s^{-1}). The Task 2 team then applied these ratios to annual average release rates of mercury from Y-12 (mg s^{-1}) for 1953 through 1962 to estimate mercury air concentrations at Scarboro for these years.

Recent measurements of mercury in tree rings of red cedars growing in the EFPC floodplain suggest that air concentrations of mercury in the floodplain were significantly elevated in the past. Because these trees are on the opposite side of Pine Ridge from the Y-12 Plant, it is assumed that much of the airborne mercury came from EFPC. Studies at other sites with elevated airborne mercury concentrations have shown that trees take up mercury from air and incorporate some of the mercury into tree rings. However, with present knowledge, mercury concentrations in individual tree rings near Oak Ridge cannot be used to reliably estimate annual average airborne mercury concentrations at the tree locations. This is because mercury is relatively mobile in the sapwood of the tree and can move from ring to ring before the sapwood becomes heartwood. In addition, individual trees appear to respond quite differently to airborne mercury.

The Task 2 team modeled volatilization of mercury from EFPC by dividing the creek into 403 straight-line segments and estimating releases rates from the creek based on assumptions about the fraction of the total mercury released from Y-12 that volatilized as the water traveled from Y-12 to the junction between EFPC and Poplar Creek. Based on mercury measurements near the junction of EFPC and Poplar Creek, information in the scientific literature, and discussions with experts at recent mercury-related conferences, the project team assumed that on average 5% of the total mercury discharged from Y-12 to EFPC in a given year escaped to the air above EFPC between Y-12 and the junction. To account for the large uncertainty in this estimate, the Task 2 team also modeled losses of 1% and 30%. The team then used the ISCST3 air dispersion model to estimate air concentrations at the *Scarboro Community*, *Robertsville School*, *EFPC Floodplain Farm Family*, and *Community Population* locations from the emissions of mercury from the creek.

Concentrations of mercury in soil and sediment used to estimate past exposures of the *Scarboro Community*, *Robertsville School Students*, *EFPC Floodplain Farm Family*, and *Community* populations to mercury in EFPC floodplain soil and EFPC sediment were estimated based on sampling conducted as part of the EFPC Floodplain Remedial Investigation during 1991 and 1992. For each population location, the Task 2 team identified samples collected from areas of the floodplain or creek likely to have been contacted by the population of interest. Concentrations reported in these samples were based on analyses of 16-inch long soil cores that were blended (or *homogenized*) prior to analysis. However, a 1992 study in which 18-inch long soil cores were divided into 1-inch depth intervals for analysis showed that, because of the frequent flooding in this area and the overlaying of more highly contaminated soils with less contaminated soils, the floodplain soil layers with the highest mercury concentrations are buried beneath up to 10 inches of soil and sediment. To adjust for the likelihood that surface soil concentrations in the past were higher than at present, and higher than the average concentrations measured in the homogenized samples, the Task 2 team applied adjustment factors determined for different time periods to the soil data sets. These subjective adjustment factors were based on the 1992 study.

Exposures of residents of the *Scarboro Community* to mercury in soil were estimated based on limited soil sampling conducted in Scarboro by Oak Ridge Associated Universities in 1984. These soils were collected to a depth of 3 inches below the surface. Soil data were not collected in Scarboro during the EFPC Floodplain Remedial Investigation. Because soils in the Scarboro Community were not subject to flooding from EFPC, adjustment factors were not applied to these data sets.

Mercury in the air and soil can be taken up by plants, and then make its way into milk and meat when cattle eat the plants. The Task 2 team derived factors to describe these transfers in the environment. The Task 2 team estimated incorporation of airborne mercury into above-ground vegetation, including fruits and vegetables and pasture grass, based on measurements of airborne mercury deposition to vegetation made over an entire year near Oak Ridge in the late 1980s. Transfer of mercury from soil to below-ground vegetables and from soil to pasture grass was estimated based on measurements of mercury in co-located soil and plant samples collected in the Oak Ridge area in the mid-1980s and in 1993. The transfer of mercury to milk and meat after intake by cattle was estimated based on studies from the literature in which soluble mercury salts were administered to cows. In estimating the transfer of mercury to milk and meat from soil ingested by cattle during grazing, the Task 2 team adjusted the transfer factors to account for the

fact that mercury in EFPC soil appears to be less available for absorption than the form administered to cows in these controlled studies.

Although it is likely that the numbers of fish in EFPC during the years of peak mercury releases from Y-12 were low due to poor water quality, anecdotal reports do suggest that a small number of individuals caught and consumed fish from EFPC during the 1950s and 1960s. The Task 2 team estimated historical annual average concentrations of mercury in EFPC fish of the size and type that may have been caught for consumption based on: 1) mercury concentrations measured in fish collected near Oak Ridge after 1970, 2) mercury concentrations measured in fish at other sites with high mercury concentrations in water and/or sediment, 3) information about the maximum possible content of mercury in fish of the size likely to have been in EFPC, 4) evidence of an upper limit of mercury concentrations found at other locations, and 5) evidence of levels of mercury that may be lethal to fish. The Task 2 team estimated ranges of historical annual average concentrations of mercury in fish from the Clinch River/ Poplar Creek and Watts Bar Reservoir that may have been caught for consumption based on mercury levels in sediment core samples collected in Poplar Creek, the Clinch River, and Watts Bar Reservoir in the mid-1980s and application of equations describing the relationship between mercury concentrations in sediment and in fish. These relationships were determined using data collected in EFPC, Poplar Creek, the Clinch River, and Watts Bar Reservoir.

There is some background exposure to mercury in the environment, because mercury is naturally present in the earth's crust and is released from a number of industrial and other man-made sources such as mining and smelting, production of chlorine gas and caustic soda, and burning of coal at coal-fired power plants (USEPA 1997). Mercury has also historically been used in a number of consumer and household products. For example, elemental mercury is used in silver-colored dental fillings, thermometers, barometers, and batteries; inorganic mercury was used widely in the past in medicinal products such as laxatives and teething powders and is still used in some fungicides, paints, and medicines; and until recently, organic mercury compounds were used as antifungal agents in some interior and exterior paints. In this assessment, exposures to mercury in some media**S** such as mercury in floodplain soil and sediment and mercury in fish**S** were evaluated primarily based on concentrations measured in the environment, and thus estimated doses reflect contributions from background exposures to mercury as well as exposures to mercury from Y-12 releases. Exposures to mercury in other media**S** such as mercury in air and mercury in EFPC water**S** were evaluated primarily based on concentration modeled to the environment based on Y-12 release data, and thus estimated doses do not explicitly reflect the contribution from background sources.

Background concentrations of mercury in the environment can range somewhat, depending on such factors as soil characteristics and proximity to former mining areas. In addition, an area's "background" concentrations can be influenced by mercury releases from distant source**S** mercury released to the atmosphere as elemental mercury vapor can be transported great distances before it is deposited at low concentrations far from its source. Table ES-2 shows ranges of background concentrations of mercury that have been measured in air, water, soil, and fish.

Table ES-2: Background Concentrations of Mercury in Different Environmental Media

Medium	Background Mercury Concentrations [a]
Air	2.0 - 10 ng m^{-3} (average US)
Water	0.001 - 0.10 µg L^{-1} (US lakes and rivers)
Soil	<0.1 - 3 mg kg^{-1} (dry weight, Eastern US soils)
Fish	0.05 - 0.1 mg kg^{-1} (fresh weight, US freshwater fish)

a ng m^{-3} = nanograms per cubic meter (a nanogram is one billionth of a gram)
µg L^{-1} = micrograms per liter (a microgram is one millionth of a gram)
mg kg^{-1} = milligrams per kilogram (a milligram is one thousandth of a gram)

Table ES-3 shows the range of mercury concentrations in air, water, soil, sediment, and fish estimated by the Task 2 team for each exposure population evaluated in this assessment for 1950-1959, 1960-1969, 1970-1979, and 1980-1990. Maximum concentrations of mercury estimated by the Task 2 team for water, soil, sediment, and fish at the locations of the exposure populations of interest exceed background concentrations for most years. Maximum concentrations of mercury estimated for air exceed background concentrations during the years when releases from Y-12 to air and water were highest.

Based on estimated concentrations in each medium of interest (air, water, soil/sediment, fruits/ vegetables, meat, milk, and/or fish) at each population location, average daily doses of mercury through all applicable exposure pathways were estimated for each year. These estimated doses, stated in terms of daily amounts of mercury taken in per kilogram of body weight (mg kg^{-1} d^{-1}), were calculated using equations that take into account the amount of air, water, soil/sediment, food, or fish that was likely inhaled, ingested, or contacted and the estimated concentrations of mercury in each of these media. The routes of exposure and chemical forms or *species* encountered in environmental exposures often differ from the toxicity studies to which they are compared. Consequently, the amounts of the chemical that are absorbed often differ even when the administered doses are the same. In the Task 2 assessment, these differences in the amount of mercury absorbed in the environmental exposures vs the toxicity studies are accounted for by *relative bioavailability* parameters. These relative bioavailability parameters are estimated based on knowledge about the forms of mercury in environmental media near Oak Ridge and in toxicity studies.

The majority of the inputs to the dose equations are not known perfectly. In many cases, there is a lack of knowledge about the true value of the parameter. For example, in the Task 2 dose reconstruction, there are incomplete records about historical operations and conditions leading to human exposure that result in uncertainties in estimated releases and model predictions. In addition, many of the parameters in the dose equations exhibit natural variability, such as personal differences in body weight and rates of food and water consumption. Some parameters may have both informational uncertainty and natural variability.

Table ES-3: Ranges of Mercury Concentrations Estimated for Each Exposure Medium [a,b]

Years	Air ($ng\ m^{-3}$)	Water ($\mu g\ L^{-1}$)	Soil ($mg\ kg^{-1}$,	Sediment ($mg\ kg^{-1}$,	Fish ($mg\ kg^{-1}$, fresh) [c]
Wolf Valley Residents					
1950-1959	0.11 - 62	---	---	---	---
1960-1969	0.13 - 10	---	---	---	---
1970-1979	---	---	---	---	---
1980-1990	---	---	---	---	---
Scarboro Community Residents					
1950-1959	0.028 - 260	2.8 - 2500	<0.1 - 3.0	<0.1 - 2900	1.5 - 4.3
1960-1969	0.015 - 26	3.6 - 270	<0.1 - 3.0	<0.1 - 1700	1.5 - 4.3
1970-1979	0.0021 - 6.1	0.66 - 64	<0.1 - 3.0	<0.1 - 580	1.3 - 3.6
1980-1990	0.0036 - 0.26	1.2 - 3.1	<0.1 - 3.0	<0.1 - 290	0.9 - 2.7
Robertsville School Students					
1950-1959	0.019 - 170	1.9 - 1800	<0.1 - 1600	<0.1 - 950	---
1960-1969	0.013 - 17	2.7 - 190	<0.1 - 1200	<0.1 - 570	---
1970-1979	0.0014 - 4.0	0.40 - 40	<0.1 - 810	<0.1 - 380	---
1980-1990	0.0024 - 0.17	0.77 - 2.0	<0.1 - 200	<0.1 - 95	---
EFPC Floodplain Farm Family					
1950-1959	0.28 - 2500	1.4- 1300	<0.1 - 580	<0.1 - 1700	1.5 - 4.3
1960-1969	0.15 - 240	2.0 - 130	<0.1 - 350	<0.1 - 990	1.5 - 4.3
1970-1979	0.020 - 58	0.27 - 27	<0.1 - 230	<0.1 - 660	1.3 - 3.6
1980-1990	0.036 - 2.4	0.52 - 1.4	<0.1 - 58	<0.1 - 170	0.9 - 2.7
Community #1					
1950-1959	0.0094 - 86	---	---	---	---
1960-1969	0.0050 - 8.3	---	---	---	---
1970-1979	0.00069 - 2.0	---	---	---	---
1980-1990	0.0012 - 0.083	---	---	---	---
Community #2					
1950-1959	0.0046 - 42	---	---	---	---
1960-1969	0.0025 - 4.0	---	---	---	---
1970-1979	0.00034 - 0.97	---	---	---	---
1980-1990	0.00059 - 0.036	---	---	---	---
Clinch River/ Poplar Creek Fish Consumers					
1950-1959	---	---	---	---	0.46 - 5.1 [d]
1960-1969	---	---	---	---	0.24 - 4.4
1970-1979	---	---	---	---	0.095 - 0.97
1980-1990	---	---	---	---	0.050 - 0.43
Watts Bar Reservoir Fish Consumers					
1950-1959	---	---	---	---	0.005 - 1.1
1960-1969	---	---	---	---	0.035 - 0.82
1970-1979	---	---	---	---	0.021 - 0.34
1980-1990	---	---	---	---	0.010 - 0.32

a Concentrations represent the 5th and 95th percentiles of the concentration distributions

b "---" indicates medium not evaluated for this population

c Concentrations represent the 5th and 95th percentiles of the *annual average* fish concentration

d Maximum fish concentrations in Clinch River/Poplar Creek are predicted to be higher than maximum concentrations in EFPC fish because it was assumed that, on average, Clinch River/Poplar Creek fish were larger

Throughout the Task 2 assessment, efforts were made to identify and quantify the uncertainty and variability in the input parameters used to estimate dose, and each parameter in the dose equations was represented by a range of values that describes what is known about the uncertainty and variability in the value of that parameter for that population. The Task 2 team then computed the total variance induced in the output by quantifying uncertainty and variability in the inputs and models using a process called probability analysis.

In probability analysis, inputs to the dose calculations are quantified not in terms of a single, discrete number, but as probability density functions (PDFs). PDFs were subjectively defined as confidence intervals within which there is a high probability of encompassing the true but unknown parameter value. The Task 2 team established PDFs based on a number of sources of information, including site-specific data, professional judgement following review of the literature, and consultation with outside experts. Whenever possible, site- or region-specific information was used, and the PDFs were based on the specific time period of interest (1950 to present).

When inputs to a dose equation are defined by distributions, each equation has many possible answers and must be solved repeatedly using different values selected from the distributions of input parameter values. In the current assessment, this process was computerized using a software program and a method known as Monte Carlo simulation. Results of the dose calculation/ uncertainty analysis process are themselves probability distributions. To reflect the overall uncertainty about the results, doses estimated in this report are stated as central estimates with 95% confidence intervals about these central estimates. The central estimates represent the most likely values based on the selected distributions for the input parameters, and the confidence intervals indicate that the investigators are 95% confident that the true dose values are no lower than the lower confidence limit and no higher than the upper confidence limit.

The results of the Task 2 reconstruction of doses can be characterized as follows:

- For all populations of interest, the highest doses were estimated to have occurred during the mid- to late-1950s. These were the years of highest releases of mercury from Y-12 to air and to EFPC.

- Excluding exposures of fish consumers to methylmercury in fish, estimated doses to the *EFPC Floodplain Farm Family* are the highest of all exposure populations that were evaluated. The estimated total dose to an EFPC Floodplain Farm Family member is dominated by consumption of fruits and vegetables contaminated from airborne mercury and inhalation of airborne mercury that volatilized from EFPC.

- Estimated total doses to *Wolf Valley ("downvalley") Residents*, resulting from direct air releases of mercury from Y-12, are also dominated by consumption of fruits and vegetables contaminated from airborne mercury. However, the highest doses estimated for this population group are about 30- to 40-times lower than the highest doses estimated for the EFPC Floodplain Farm Family.

- Estimated total doses to *Scarboro Community Residents* are dominated by consumption of fruits and vegetables contaminated from airborne mercury, incidental ingestion of waterborne mercury, skin contact with contaminated EFPC water and sediment, and inhalation of airborne mercury due to both direct air releases of mercury from Y-12 and volatilization of mercury from EFPC. The highest estimated inhalation doses (estimated for 1955) are about 9-times lower than the highest inhalation doses estimated for the EFPC Floodplain Farm Family (estimated for 1957), due in part to the greater distance of the Scarboro Community from EFPC.

- Estimated total doses to *Robertsville School Students* are dominated by incidental ingestion of and skin contact with mercury in floodplain soil and in EFPC water.

- Estimated doses to *Community Populations 1* and *2*, for which exposures from airborne mercury volatilized from EFPC were evaluated, were comprised of inhalation of airborne mercury and consumption of fruits and vegetables contaminated from airborne mercury only.

- Estimated methylmercury doses to *Fish Consumers* who consumed fish from Watts Bar Reservoir were about 4-fold lower than doses estimated for *Fish Consumers* who consumed the same amount of fish from Clinch River/ Poplar Creek.

In order to put the Task 2 dose estimates in perspective and evaluate the likelihood that the estimated levels of historical exposure caused adverse health effects, the project team collected, evaluated, and summarized available studies of the toxicity of different species of mercury through various routes of exposure. The following sources of information were evaluated to address the potential for toxic effects from ingestion of inorganic mercury, inhalation of elemental mercury, and ingestion of methylmercury:

- USEPA and ATSDR recommended levels of concern for exposure to inhaled elemental mercury vapor, ingested inorganic mercury, and ingested methylmercury (that is, EPA's "reference doses" and ATSDR's "minimal risk levels").

- Worker exposure studies that investigated evidence of adverse health effects in workers exposed to airborne elemental mercury vapor for many years.

- Studies that investigated evidence of no observable adverse effects levels (NOAELs) and lowest observable adverse effects levels (LOAELs) in laboratory animals that were administered different doses of inorganic mercury or that inhaled different concentrations of elemental mercury vapor.

- Information from exposures of people to mercury in fish from Minamata Bay and Niigata, Japan following releases of methylmercury from a chemical manufacturing plant.

- Information from exposures of people in Iraq to methylmercury in treated seed grain that was used to make bread.

- Information from studies of "fish-eating" populations that consume large quantities of fish containing lower levels of methylmercury, including studies in the Seychelles Islands, northern Quebec, and New Zealand.

Using these sources of information, the Task 2 team established toxicity benchmark values for comparison with doses estimated in the dose reconstruction. Table ES-4 shows the toxicity benchmark values used in this report.

Table ES-4: Toxicity Benchmarks Values for Comparison with Results of the Mercury Dose Reconstruction

Species and Exposure Route	No Observed Adverse Effect Level ($mg\ kg^{-1}\ d^{-1}$)	USEPA Reference Dose[a] ($mg\ kg^{-1}\ d^{-1}$)	ATSDR Minimal Risk Level[b] ($mg\ kg^{-1}\ d^{-1}$)
Inhalation of Elemental Mercury	0.0029 to 0.0071 (human studies)[c]	0.000086	0.000057
Ingestion of Inorganic Mercury	0.1 to 0.23 (animal studies)[d]	0.0003	0.002[e]
Ingestion of Methylmercury— *In utero* and child exposure	0.0005 (human studies)[c]	0.0001	0.0005[f]
Ingestion of Methylmercury— Adult exposure	NA	0.0003	NA

a Reference: IRIS 1998, USEPA 1985 (adult methylmercury RfD)
b Reference: ATSDR 1997.
c Data points given are from studies in humans.
d Minimal data are available. Data points given are from studies in laboratory animals.
e For intermediate duration exposures.
f For chronic duration exposures.
NA Not available.

The primary toxicity benchmark values used in this report are USEPA references doses (RfDs), Lowest or No Observed Adverse Effects Levels (LOAELs or NOAELs), and ATSDR Minimal Risk Levels (MRLs). RfDs are regulatory levels below which it is unlikely that a dose will be associated with adverse health effects, given the safety criteria built into these criteria, but above which a dose may need to be investigated further to evaluate the likelihood of a health effect. LOAELs are the *lowest* doses at which adverse health effects *were* observed, while NOAELs are the *highest* doses at which adverse health

effects *were not* observed. RfDs and MRLs are derived from LOAELs and NOAELs by dividing them by safety factors that can range from 10 to 10,000. Although adverse health effects are observed at or near LOAELs and NOAELs, in many cases the effects were observed in animal studies (for example, the LOAELs and NOAELs for inorganic mercury are based on studies in rodents), in occupational exposures (for example, the LOAELs and NOAELs for elemental mercury are based on worker exposures to airborne mercury vapor), or may not be based on the most recent scientific studies due to the lengthy regulatory review process (for example, recently published LOAELs and NOAELs for fish-consuming populations exposed to methylmercury in fish are higher than earlier values).

Figures ES-3, ES-6, and ES-13 show the years that the 95% subjective confidence interval of the estimated doses of elemental mercury, inorganic mercury, and methylmercury, respectively, exceeded the applicable USEPA RfD and NOAELs at the *upper bound* (97.5^{th} percentile), *central value* (50^{th} percentile), and *lower bound* (2.5^{th} percentile) of the annual average dose. Elemental mercury doses were assumed to be comprised of exposures from inhalation of airborne mercury vapor. Methylmercury doses were assumed to be comprised of exposures from consumption of mercury in fish. Total inorganic mercury doses were assumed to be comprised of exposures from all of the remaining pathways, including ingestion and skin contact with EFPC water and sediment, ingestion and skin contact with soil, and consumption of milk, meat, and fruits and vegetables. For the years that the estimated annual average elemental, total inorganic, and methylmercury doses at the *upper bound* (97.5^{th} percentile) of the 95% subjective confidence interval are less than the corresponding USEPA RfD, it is *not likely* that adverse health effects occurred as a result of historical exposures to mercury from the Y-12 Plant during these years, based on current scientific knowledge. Since a hazard index is defined as the ratio of a dose to the applicable RfD, exceeding the RfD is equivalent to exceeding a hazard index of 1.

Figures ES-4, ES-5, ES-7, ES-8, and ES-14 show how the highest estimated doses of each of the three forms of mercury for children and adults in each population compare to the RfDs and the NOAELs. For each population, the highest doses were estimated between 1955 and 1958. For the Wolf Valley Residents, the highest doses were estimated for 1955, because this was the year that the estimated airborne releases from Y-12 were highest. For the Scarboro Community Residents population, the highest doses were also estimated for 1955, because air concentrations at Scarboro (assumed to come from *both* direct airborne releases from Y-12 and volatilization of mercury from EFPC) were estimated to be highest during this year. Estimated inorganic mercury doses to Scarboro Community Residents during 1955 were slightly higher than inorganic mercury doses estimated for 1957 and 1958, the years of highest mercury releases to EFPC, because total inorganic mercury doses were estimated to be dominated by consumption of homegrown fruits and vegetables contaminated by airborne elemental mercury (the airborne mercury was assumed to be incorporated into the plant as inorganic mercury). For the remaining populations, the highest doses were estimated for 1957 and 1958, because these were the years of highest mercury releases to EFPC.

The following general conclusions can be drawn from the Task 2 mercury dose reconstruction based on the estimated annual-average doses shown in the figures:

Inhalation of airborne (elemental) mercury:

(Figure ES-3 highlights the years that estimated elemental mercury doses from inhalation exceeded the elemental mercury RfD and the years that estimated doses exceeded the NOAEL, and Figures ES-4 and ES-5 compare the highest estimated doses for children and adults, respectively, in each population to the RfD and the NOAEL):

- *Comparison to RfDs***S**The 95% confidence interval on the estimated inhalation doses of elemental mercury exceeded the RfD at two population locations: the *Scarboro Community* for 1955, 1957, and 1958 (child) and the *EFPC Floodplain Farm Family* location for 1953-1960 (child) and 1955-1959 (adult).

- *Comparison to NOAELs***S** The 95% subjective confidence intervals on the estimated annual average elemental mercury doses for *all* populations and *all* years were below the NOAELs. These NOAELs were established from studies of workers exposed to airborne mercury vapor for prolonged periods of time. Neurological effects, including hand tremor, increases in memory disturbances, and evidence of dysfunction of the autonomic (involuntary) nervous system (IRIS 1998) were reported in some workers exposed at doses above the NOAELs. At slightly higher doses, evidence of effects on the kidney have also been observed. The USEPA RfD is about 30 times lower than the NOAEL because it incorporates a conservative safety factor. Health effects in humans exposed to elemental mercury at doses at or below the NOAEL have not been reported.

- *Populations with the highest exposures***S** The highest estimated elemental mercury doses were to children who were members of the *EFPC Floodplain Farm Family* in 1957. The *upper bound* on the highest estimated annual average elemental mercury inhalation dose (0.0011 mg kg d^{-1} for the *EFPC Floodplain Farm Family* child in 1957) is about 13-times higher than the RfD derived from USEPA's reference concentration, but about 1/3 of the NOAEL. The *upper bound* estimates of inhalation doses are based on uncertain estimates of airborne mercury concentrations from transport of Y-12 airborne emissions over Pine Ridge and emission of elemental mercury from the waters of EFPC.

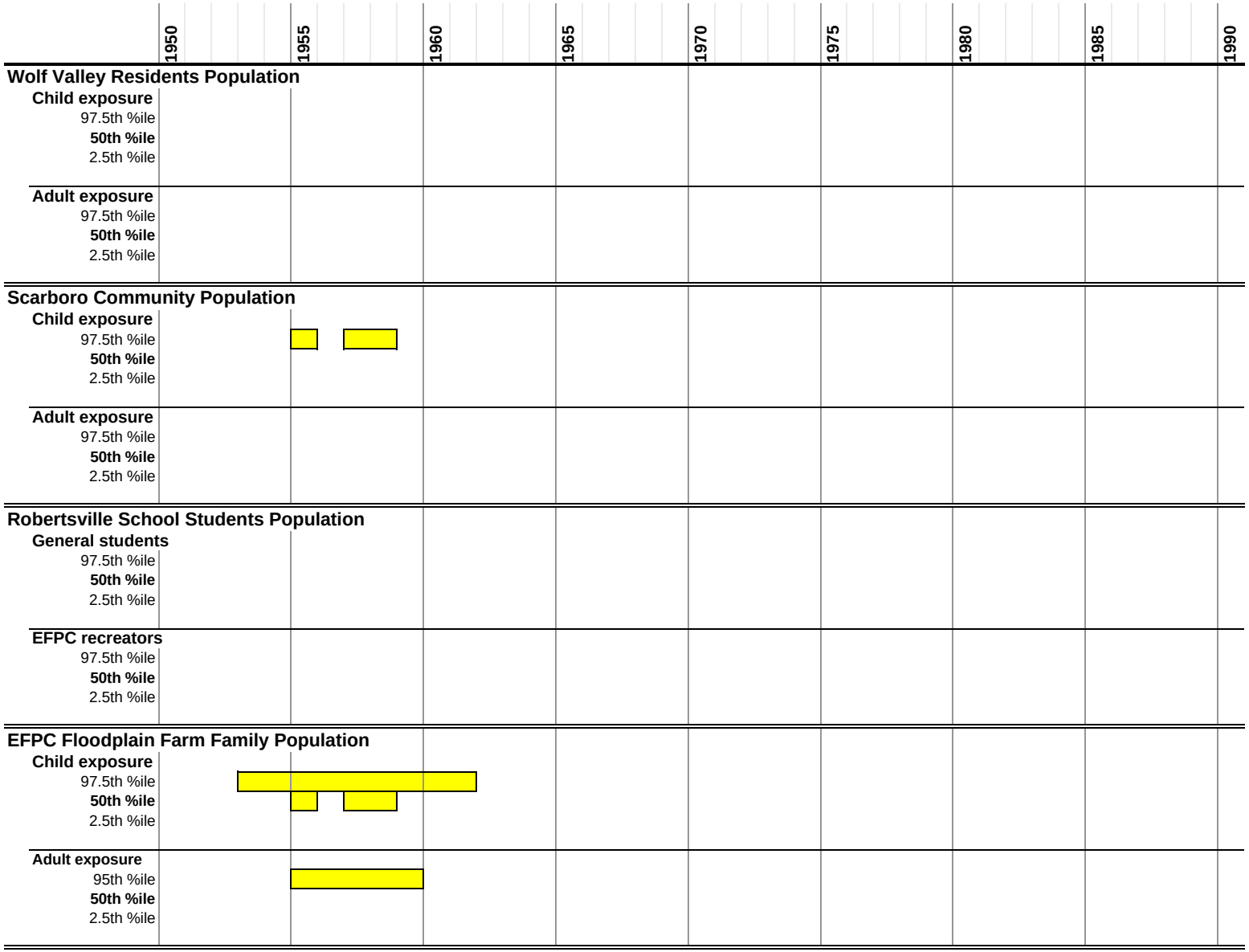
  Estimated doses from inhalation for the *Scarboro Community* population during 1953-1962 (when air concentrations at this location were assumed to result from both direct airborne mercury releases from Y-12 that were transported over Pine Ridge, and volatilization of mercury from EFPC) are about 15% to 40% of the inhalation doses estimated for the *EFPC Floodplain Farm Family* population during these years. During other years, estimated doses at Scarboro are about 10% of doses estimated at the EFPC Floodplain Farm Family location. The

higher estimated doses at the EFPC Floodplain Farm Family location are due to its closer proximity to EFPC.

- *Likelihood of exposures above the RfD, Scarboro Community Residents***S** The estimated size of the *Scarboro Community* population was assumed to be between 800 and 1,200 individuals per year. Since estimated doses at the 50th percentile for this population were below the RfD for all years, it is likely that doses to most individuals in this population were below the RfD. However, because of the relatively large size of this population, it is possible that inhalation doses to a small number of people in this population during the years of highest mercury releases from Y-12 (1953-1962) may have exceeded the RfD.

- *Likelihood of exposures above the RfD, EFPC Floodplain Farm Family members***S** The estimated size of the EFPC Floodplain Farm Family population was very small (a total of between 10 and 50 individuals were assumed in this population per year). Since estimated doses at the 50th percentile to some members of this population exceeded the RfD during the years of highest mercury releases from Y-12, it is likely that doses to some individuals in this population exceeded the RfD.

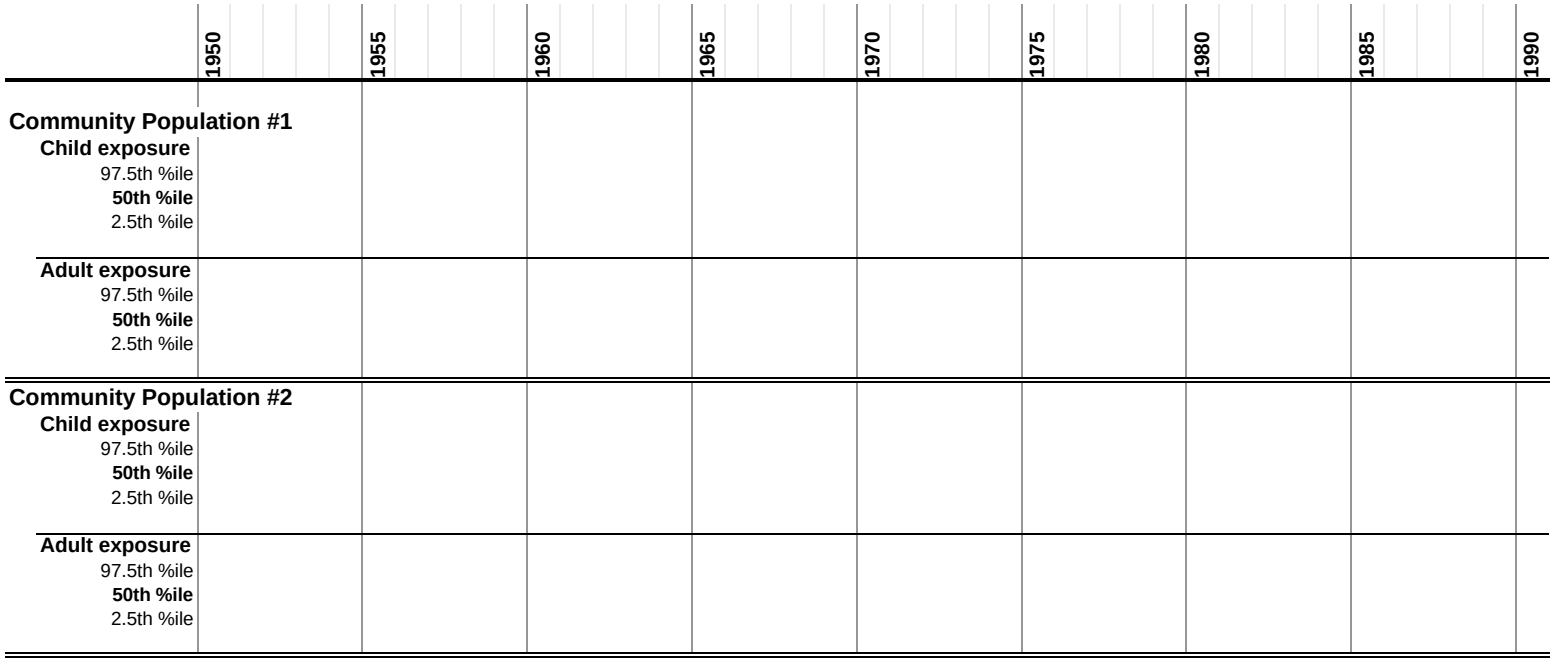
Figure ES-3: Years that the Estimated Elemental Mercury Doses from Inhalation Exceeded the USEPA RfD and the NOAEL (page 1 of 2) [a]



a Years that exceeded the RfD are indicated with light shading (USEPA RfD = 0.000086 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL (if any) are indicated with dark shading (NOAEL = 0.0029 mg kg^{-1} d^{-1})

(None on this page)

Figure ES-3: Years that the Estimated Elemental Mercury Doses from Inhalation Exceeded the USEPA RfD and the NOAEL (page 2 of 2) [a]



a Years that exceeded the RfD are indicated with light shading (USEPA RfD = 0.000086 mg kg^{-1} d^{-1}) (None on this page)
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0029 mg kg^{-1} d^{-1}) (None on this page)

Figure ES-4: Elemental Mercury (Child exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values

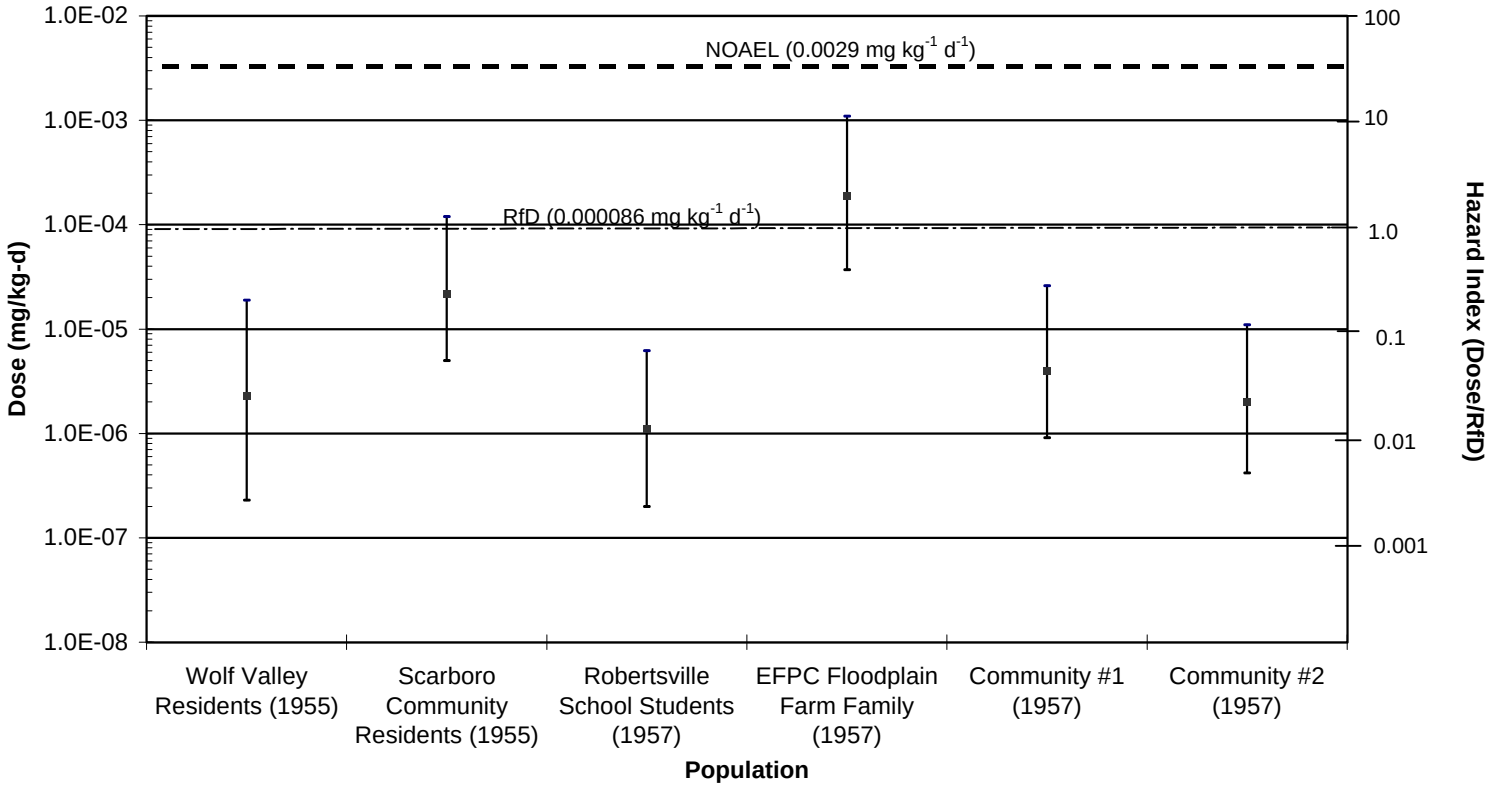
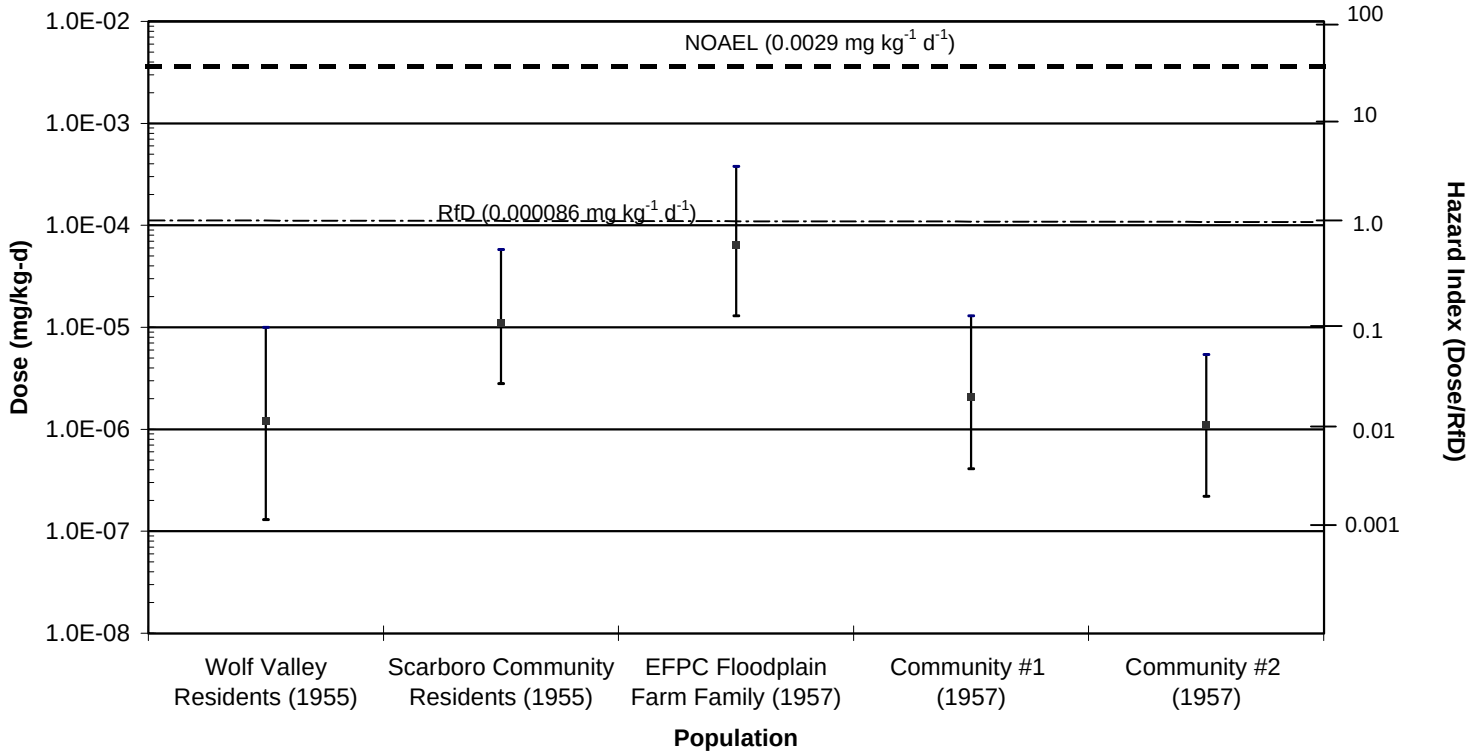


Figure ES-5: Elemental Mercury (Adult exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values



Ingestion and contact with inorganic mercury in soil, sediment, water, meat, milk, and fruits/ vegetables:

(Figure ES-6 highlights the years that estimated inorganic mercury doses exceeded the inorganic mercury RfD and the years that the estimated doses exceeded the NOAEL; Figures ES-7 and ES-8 compare the highest estimated doses for children and adults, respectively, in each population to the RfD and the NOAEL; and Figures ES-9, ES-10, ES-11, and ES-12 show which pathways contributed the most to the highest estimated total inorganic mercury doses for each population):

- *Comparison to RfDs***S** The 95% subjective confidence interval on estimated inorganic mercury doses exceeded the USEPA RfD for inorganic mercury for at least one year for all six non-angler populations evaluated in this assessment: *Wolf Valley Residents* (child**S** 1955), the *Scarboro Community* (child**S** 1953-1962, adult**S** 1954-1959), *Robertsville School Students* (general student**S** 1955-1956, 1958; recreator**S** 1955-1958), the *EFPC Floodplain Farm Family* (child**S** 1950-1970, 1973, adult**S** 1952-1965), and the two *Oak Ridge Community* populations (Community Population #1 child**S** 1955, 1957-1958, Community Population #2 child**S** 1958).

- *Comparison to NOAELs***S** The 95% subjective confidence interval on estimated annual average inorganic mercury doses for *all* populations and *all* years were below the NOAEL for inorganic mercury. The NOAEL for inorganic mercury is based on kidney effects observed in rats fed high concentrations of water soluble mercuric chloride. The USEPA RfD is about 1,000 thousand to 3,000 times lower than reported NOAELs, because it incorporates a conservative margin of safety to account for the lack of data on the toxicity of inorganic mercury to humans. Health effects in humans exposed to inorganic mercury at doses at or below the NOAEL have not been reported.

- *Populations with the highest exposures***S** The highest estimated inorganic mercury doses were to children who were members of the *EFPC Floodplain Farm Family* in 1958. The *upper bound* on the highest estimated annual average inorganic mercury dose (0.027 mg kg d^{-1} for the *EFPC Floodplain Farm Family* child in 1958) is about 90-times higher than the USEPA RfD, but about 1/4 of the NOAEL. Doses to these individuals were estimated to be high because they were assumed to live close to EFPC on the edge of the floodplain and to be exposed through multiple pathways, including contact with contaminated soil, sediment, and water, and ingestion of "backyard" fruits/vegetables, milk, and meat. Inorganic mercury doses to *Scarboro Community Residents* during the mid- 1950s to early-1960s were also estimated to potentially exceed the RfD, because it was assumed that they occasionally recreated in EFPC (at a location only about 1 to 1½-mile downstream of Y-12) and consumed "backyard" fruits/vegetables.

- *Important pathways*— Estimated total inorganic mercury doses exceeded the RfD at all six locations. At five of the six locations, estimated doses were largely contributed by ingestion of homegrown fruits and vegetables contaminated by airborne mercury. This pathway was not evaluated for the *Robertsville School Students;* for this population, exposures were dominated by contact with contaminated surface soil and contact with contaminated water in EFPC. Contact with contaminated water in EFPC was the second most important pathway for *Scarboro Community Residents.* Contact with contaminated surface soil was the second most important pathway for *EFPC Floodplain Farm Family* members.

- *Likelihood of exposures above the RfD, Wolf Valley Residents***S** The estimated size of the *Wolf Valley Residents* population was small (between 30 to 100 people in a given year). For this population, the results of this assessment suggest that doses to young children *only* may have exceeded the RfD, and only if they consumed very large quantities of homegrown above-ground fruits and vegetables. Because of the small size of this population and the relatively low doses estimated for them, it is likely that the number of individuals in this population who were exposed to inorganic mercury at doses above the RfD, if any, was small.

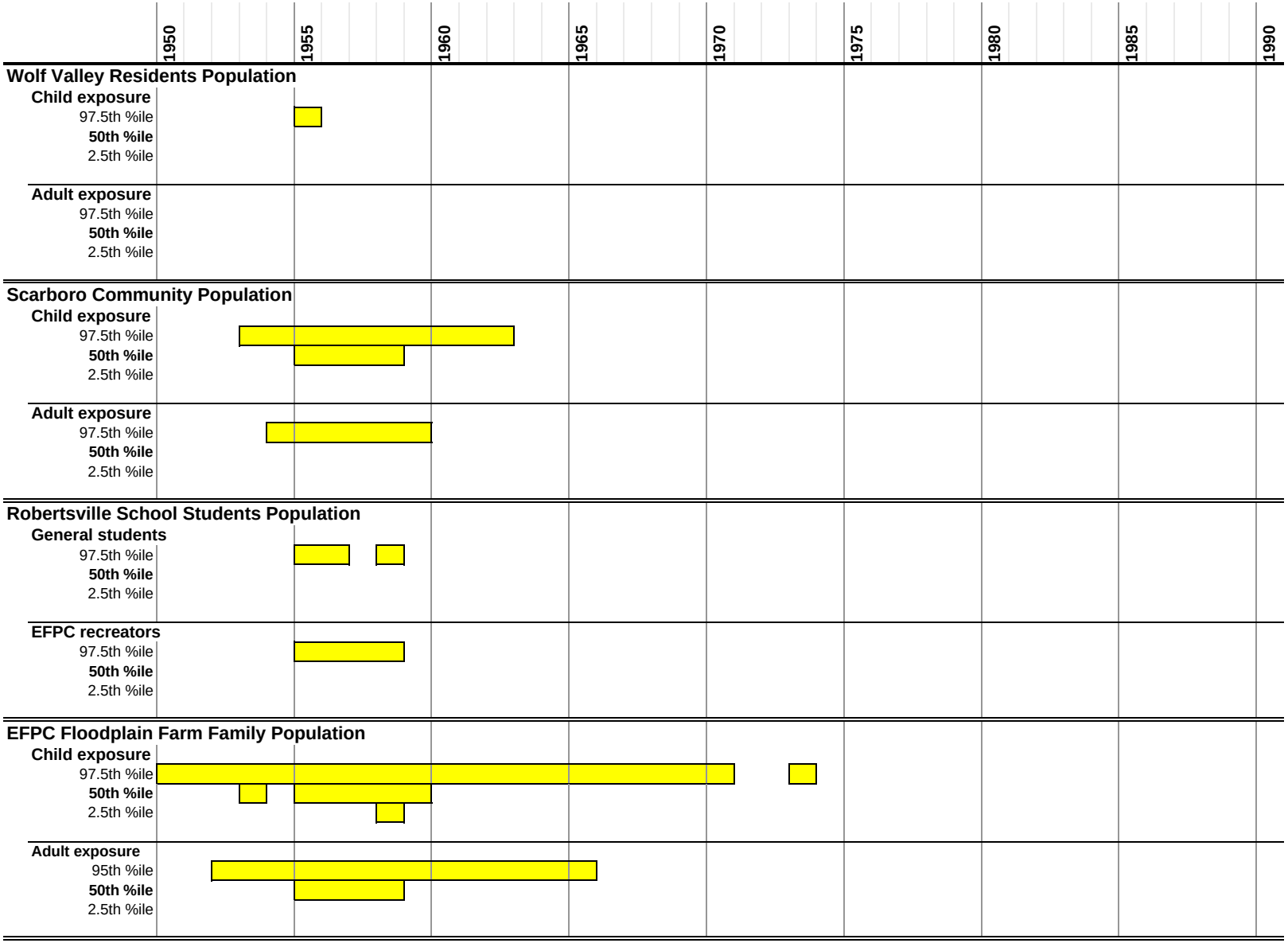
- *Likelihood of exposures above the RfD, Scarboro Community Residents***S** The estimated size of the *Scarboro Community Residents* population was relatively large (between 800 and 1,200 individuals in a given year). Since estimated doses at the 50th percentile for this population were below the RfD for most years, it is likely that doses to most individuals in this population were below the RfD. However, because of the relatively large size of this population, it is possible that inorganic mercury doses to a moderate number of people in this population during the years of highest mercury releases from Y-12 (1953-1962) may have exceeded the RfD, particularly for those individuals who frequently recreated in EFPC or regularly consumed above-ground fruits/vegetables from backyard gardens.

- *Likelihood of exposures above the RfD, Robertsville School Students***S** The estimated size of the *Robertsville School* general student population was relatively large (between 1,500 and 2,000 students in a given year). Since estimated doses at the 50th percentile for this population were below the RfD for all years, and doses at the 97.5th percentile exceeded the RfD only during a few years in the mid-1950s, it is likely that the number of individuals in this population who were exposed to inorganic mercury at doses above the RfD was small. Behaviors most likely to have resulted in doses above the RfD were frequent contact with schoolyard soil, particularly near EFPC, and frequent contact with EFPC water and sediment.

- *Likelihood of exposures above the RfD, EFPC Floodplain Farm Family members***S** The estimated size of the *EFPC Floodplain Farm Family* population was very small (between 10 and 50 individuals in a given year). Because estimated doses at the 50th percentile for this population exceeded the RfD during the years of highest mercury releases from Y-12 (1953-1962) and because this population group was assumed to live close to EFPC, it is likely that doses to some individuals in this population exceeded the RfD. Behaviors most likely to have resulted in doses above the RfD were frequent contact with floodplain soil and EFPC water and sediment, and consumption of "backyard" fruits and vegetables.

- *Likelihood of exposures above the RfD, Community Populations***S** The estimated size of the *Community Populations* was relatively large (between 1,500 and 2,000 individuals in a given year). However, the results of this assessment suggest that for these populations, doses to young children *only* may have exceeded the RfD if they consumed very large quantities of homegrown above-ground fruits and vegetables during the years of highest mercury releases from Y-12 (mid-1950s) *and* lived closer than one-mile to the creek. Consequently, it is likely that the number of individuals in these populations who were exposed to inorganic mercury at doses above the RfD, if any, was small.

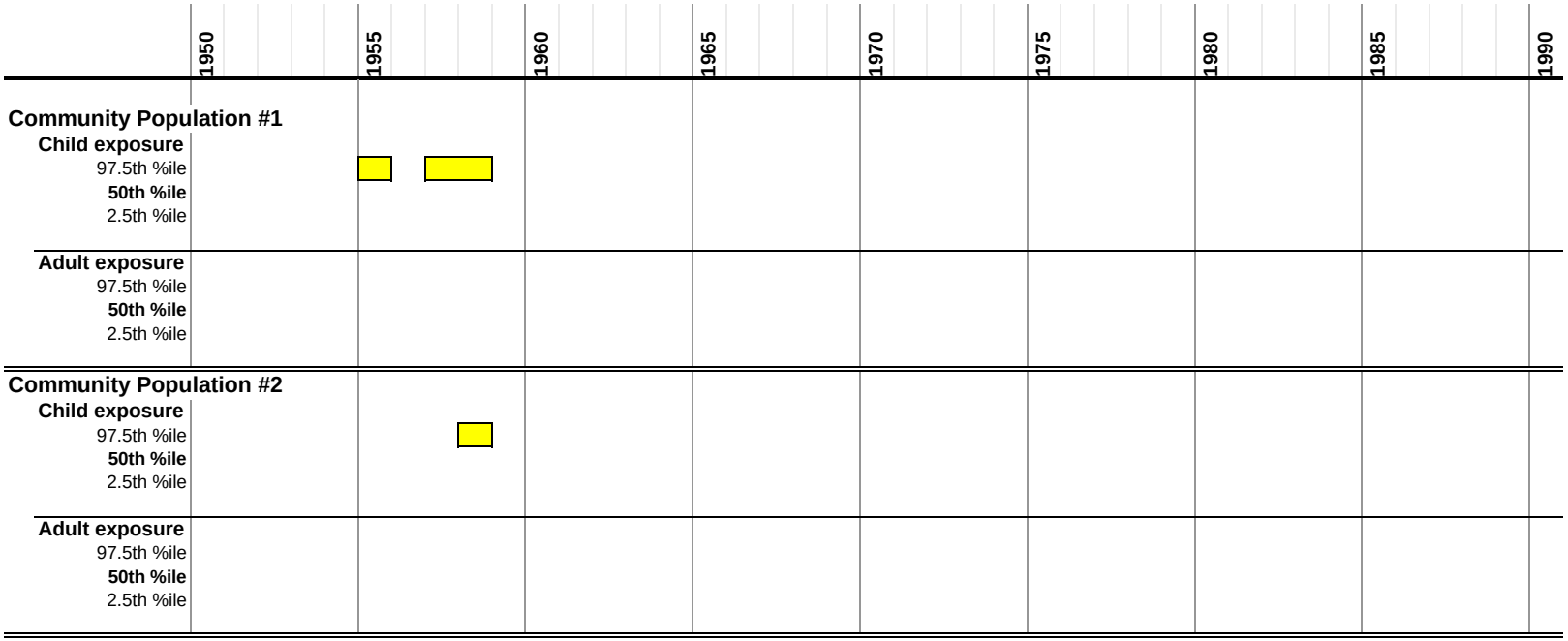
Figure ES-6: Years that the Estimated Total Inorganic Mercury Doses (from All Pathways Except Inhalation and Fish Consumption) Exceeded the USEPA RfD and the NOAEL (page 1 of 2) [a]



a Years that exceeded the RfD are indicated with light shading (USEPA RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.1 mg kg^{-1} d^{-1})

(None on this page)

Figure ES-6: Years that the Estimated Total Inorganic Mercury Doses (from All Pathways Except Inhalation and Fish Consumption) Exceeded the USEPA RfD and the NOAEL (page 2 of 2) [a]



a Years that exceeded the RfD are indicated with light shading (USEPA RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.1 mg kg^{-1} d^{-1})

(None on this page)

Figure ES-7: Inorganic Mercury (Child exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values

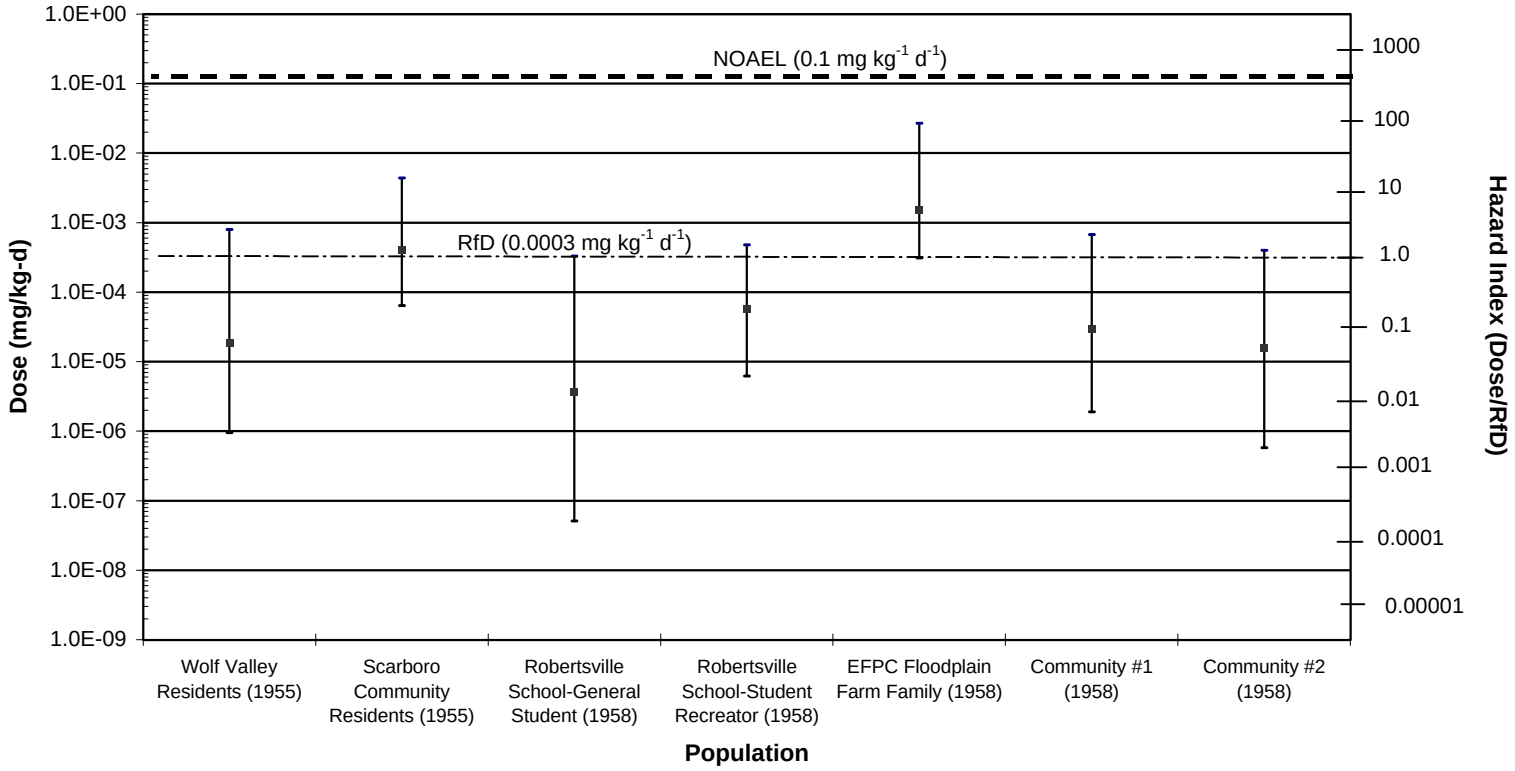


Figure ES-8: Inorganic Mercury (Adult exposure)-
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

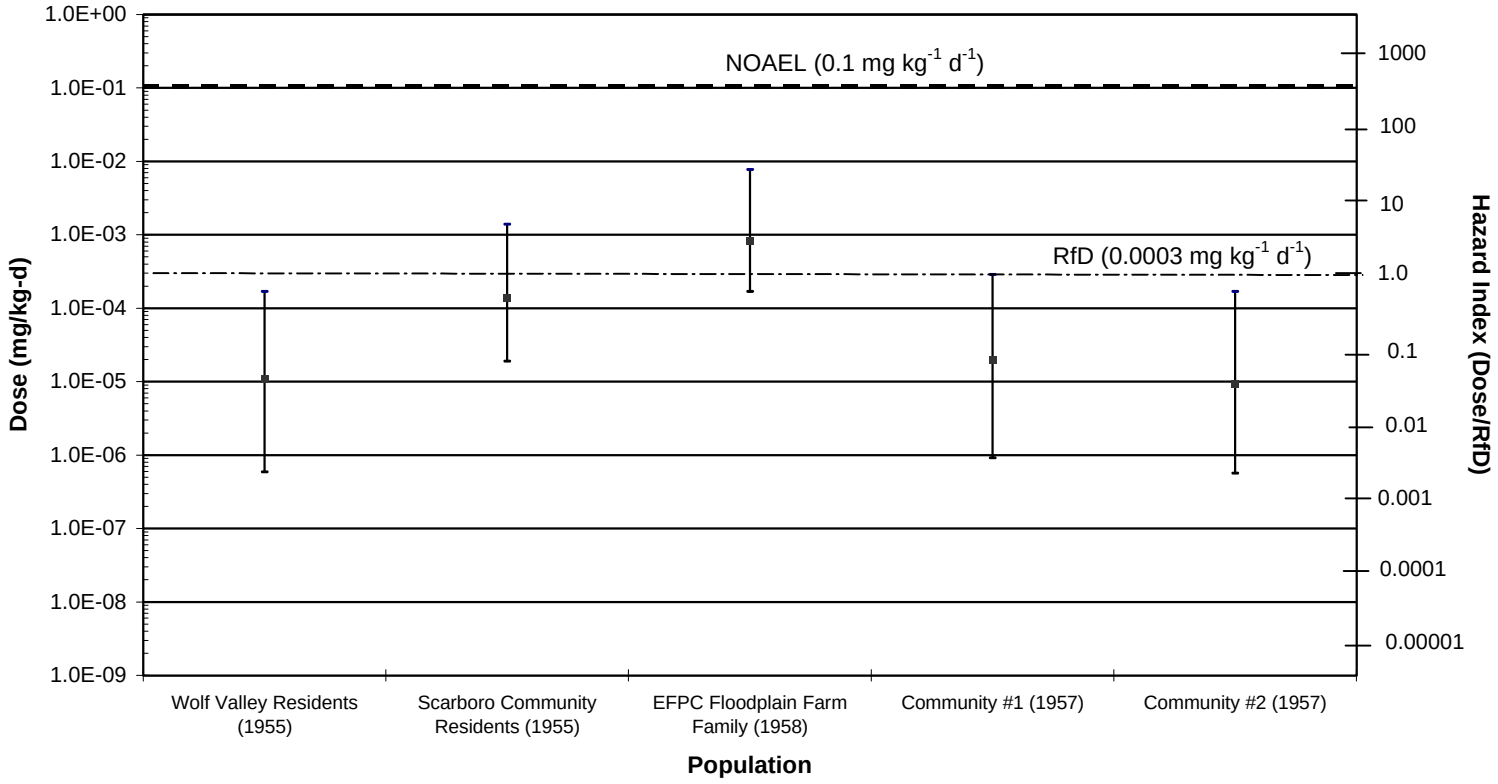


Figure ES-9: Contribution of Individual Pathways to the Estimated Total Inorganic Mercury Dose (based on the average estimated dose for the highest year)

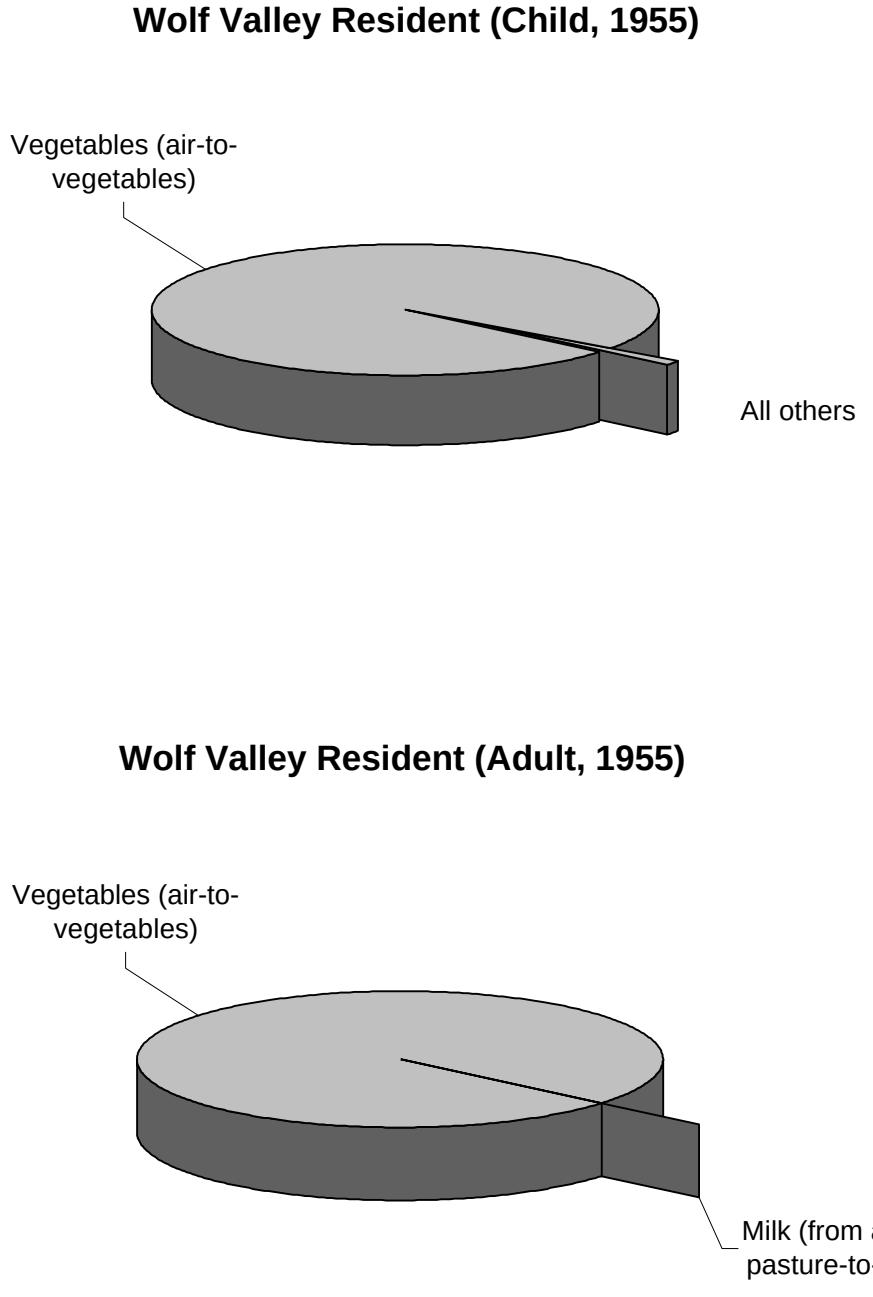
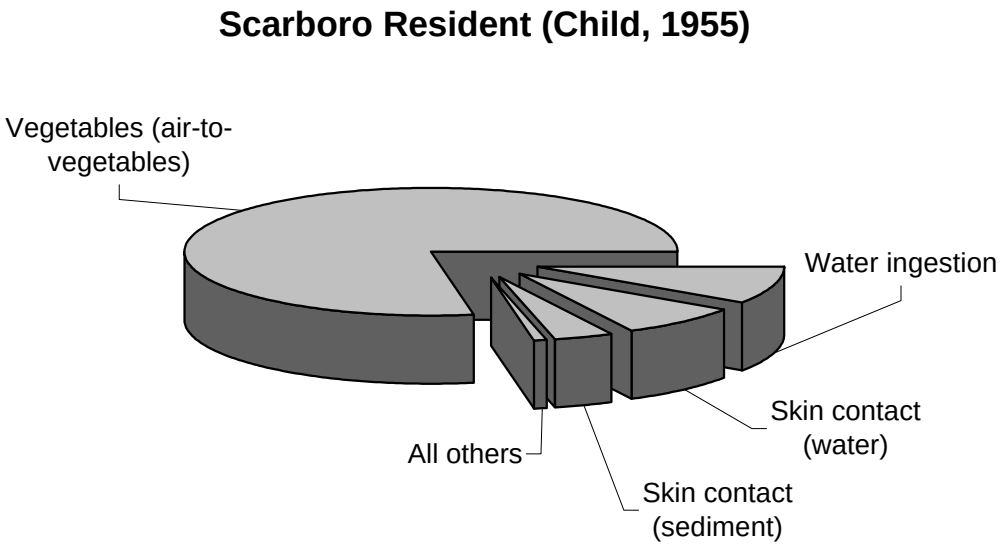


Figure ES-10: Contribution of Individual Pathways to the Estimated Total Inorganic Mercury Dose (based on the average estimated dose for the highest year)



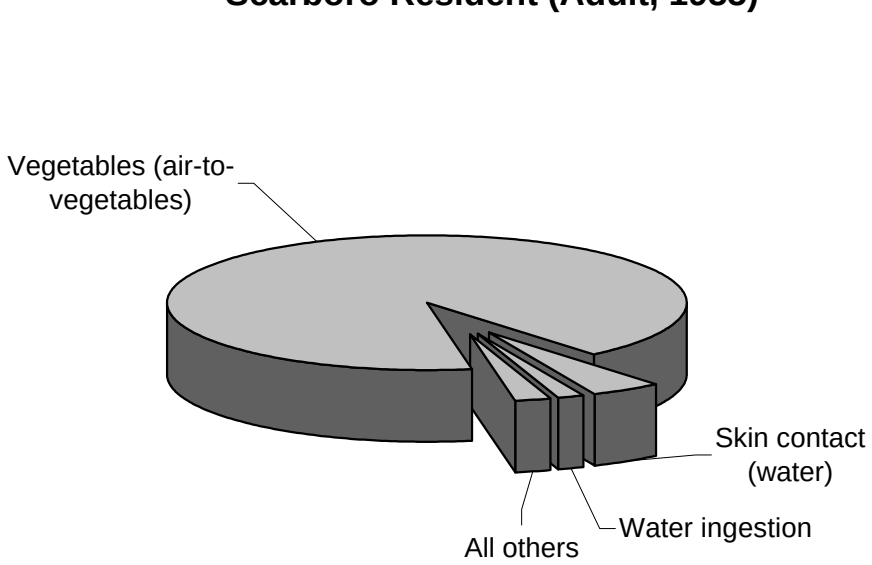


Figure ES-11: Contribution of Individual Pathways to the Estimated Total Inorganic Mercury Dose (based on the average estimated dose for the highest year)

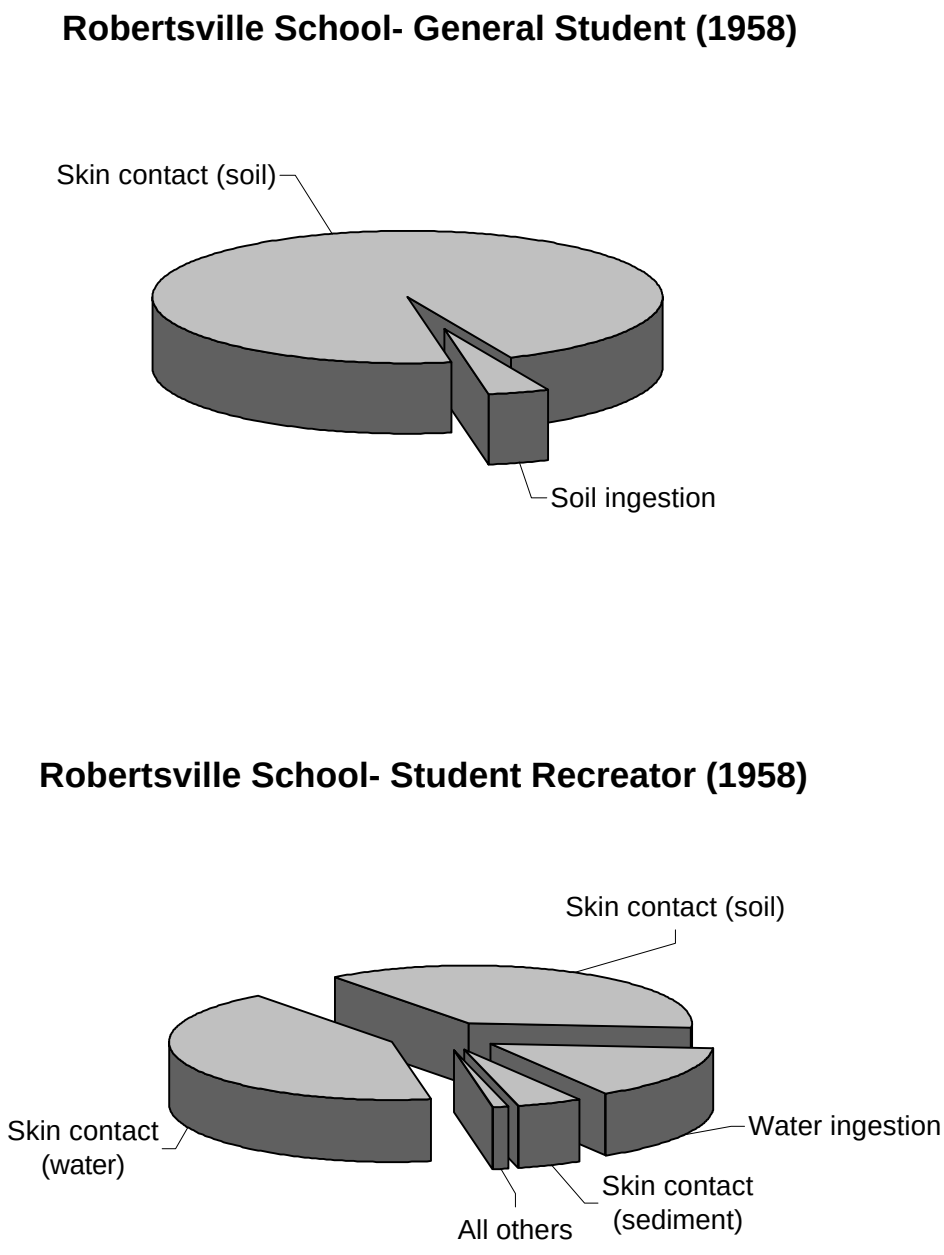
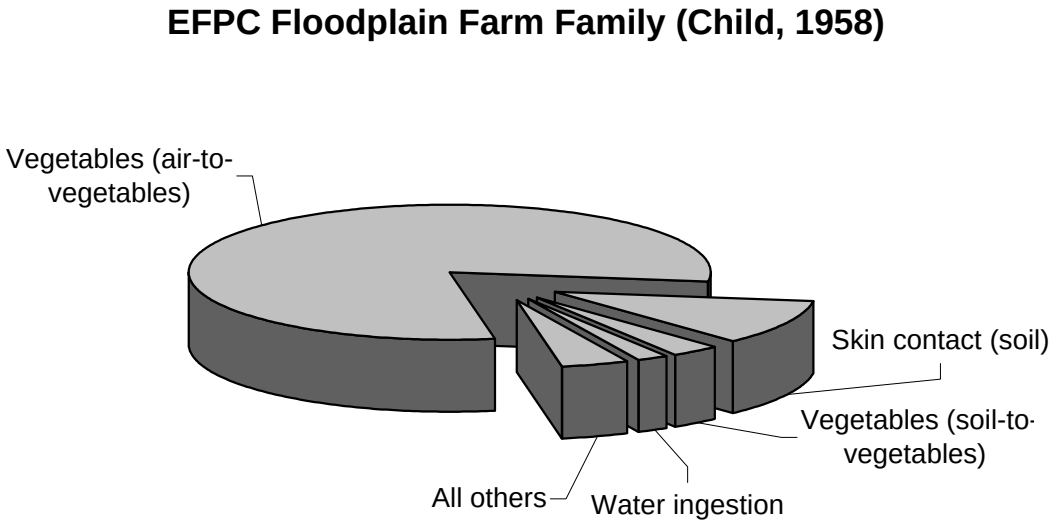
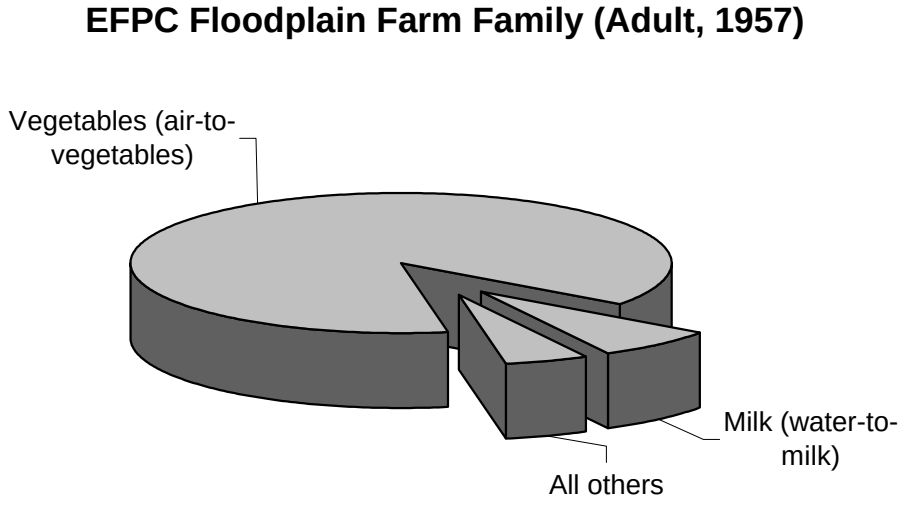


Figure ES-12: Contribution of Individual Pathways to the Estimated Total Inorganic Mercury Dose (based on the average estimated dose for the highest year)





Ingestion of methylmercury in fish:

(Figure ES-13 highlight the years that estimated methylmercury doses from consumption of fish exceeded the methylmercury RfDs and the years that estimated doses exceeded the NOAEL and Figure ES-14 compares the highest estimated doses in each population to the RfD and the NOAEL):

*Consumers of Fish from Watts Bar Reservoir***S**

- *Comparison to RfDs***S** The 95% subjective confidence interval on estimated methylmercury doses from consumption of fish exceeded the USEPA RfD based on *in utero* exposures for all years for Category 1 fish consumers, 1950-1980 for Category 2 fish consumers, and 1957-1959 for Category 3 fish consumers. During the years of highest mercury releases from Y-12 (1956-1960), estimated doses for Category 1 fish consumers exceeded the RfD based on *in utero* exposures even at the *lower bound* of the distribution (the 2.5th percentile).

- *Comparison to NOAELs***S** The 95% subjective confidence interval on estimated methylmercury doses exceeded the NOAEL for 1956-1960 for Category 1 fish consumers. Estimated doses to Category 2 and 3 fish consumers were below the NOAEL. The NOAEL for methylmercury is based on observations of neurological effects in children who were exposed to methylmercury *in utero* when their mothers consumed methylmercury in fish during pregnancy. Health effects in humans exposed to methylmercury at doses at or below the NOAEL have not been reported.

- *Exposures to children***S** Based on calculations by the Task 2 team, children who ate as few as 3 to 4 meals of fish from Watts Bar Reservoir during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate 7 or more meals of fish per year from Watt Bar Reservoir during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

- *Exposures to adults***S** Based on calculations by the Task 2 team, adults who ate 9 or more meals of fish from Watts Bar Reservoir during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate about 20 or more meals per year during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD. Adults who were not pregnant could have consumed about three times as many fish meals per year as pregnant adult females, without being at risk of adverse health effects from methylmercury exposure, because it is believed that adults are not as sensitive to adverse health effects from

methylmercury exposure as children who were exposed *in utero*. The estimated number of fetuses placed at risk is uncertain, but is nearer to 100 than to 1,000.

- *Likelihood of exposures above the RfD***S** The estimated size of the recreational angler population in Watts Bar Reservoir was large (between 10,000 and 30,000 individuals in a given year). Because Watts Bar Reservoir was a productive and popular recreational fishery, it is likely that a significant number of people annually consumed a large number of fish from this system and, particularly during the mid-1950s and 1960s, were exposed to methylmercury at doses that exceeded the USEPA RfD.

*Consumers of Fish from Clinch River/ Poplar Creek***S**

- *Comparison to RfDs***S** The 95% subjective confidence interval on estimated methylmercury doses from consumption of fish exceeded the USEPA RfD based on *in utero* exposures for all years for Category 1 fish consumers, 1950-1982 for Category 2 fish consumers, and 1950-1966 for Category 3 fish consumers. Estimated doses exceeded the RfD based on *in utero* exposures even at the *lower bound* of the distribution (the 2.5th percentile) for 1950-1975 for Category 1 fish consumers and 1950-1964 for Category 2 fish consumers.

- *Comparison to NOAELs***S** The 95% subjective confidence interval on estimated methylmercury doses exceeded the NOAEL for 1950-1975 for Category 1 fish consumers, 1950-1964 for Category 2 fish consumers, and 1957 for Category 3 fish consumers. The NOAEL for methylmercury is based on observations of neurological effects in children who were exposed to methylmercury *in utero* when their mothers consumed methylmercury in fish during pregnancy. Health effects in humans exposed to methylmercury at doses at or below the NOAEL have not been reported.

- *Exposures to children***S** Based on calculations by the Task 2 team, children who ate as few as 1 meal of fish from Clinch River/ Poplar Creek during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate about 2 or more meals of fish per year from Clinch River/Poplar Creek during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

- *Exposures to adults***S** Based on calculations by the Task 2 team, adults who ate 2 to 3 or more meals of fish from Clinch River/Poplar Creek during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate 5 or more meals per year during these years, it is *likely* that they were exposed to methylmercury at doses

that exceeded the USEPA RfD. Adults who were not pregnant could have consumed about three times as many fish meals per year as pregnant adult females, without being at risk of adverse health effects from methymercury exposure because it is believed that adults are not as sensitive to adverse health effects from methylmercury exposure as children who were exposed *in utero*.

- *Likelihood of exposures above the RfD***S** The estimated size of the recreational angler population in Clinch River/Poplar Creek was large (between 3,000 and 10,000 individuals in a given year). Because a large number of people occasionally fished in Clinch River/Poplar Creek and many likely consumed moderate quantities of fish from this system, it is likely that a significant number of people who caught and consumed fish from this system were exposed to methylmercury at doses that exceeded the USEPA RfD, particularly if they consumed fish from this system during the mid-1950s and 1960s.

*Consumers of Fish from EFPC***S**

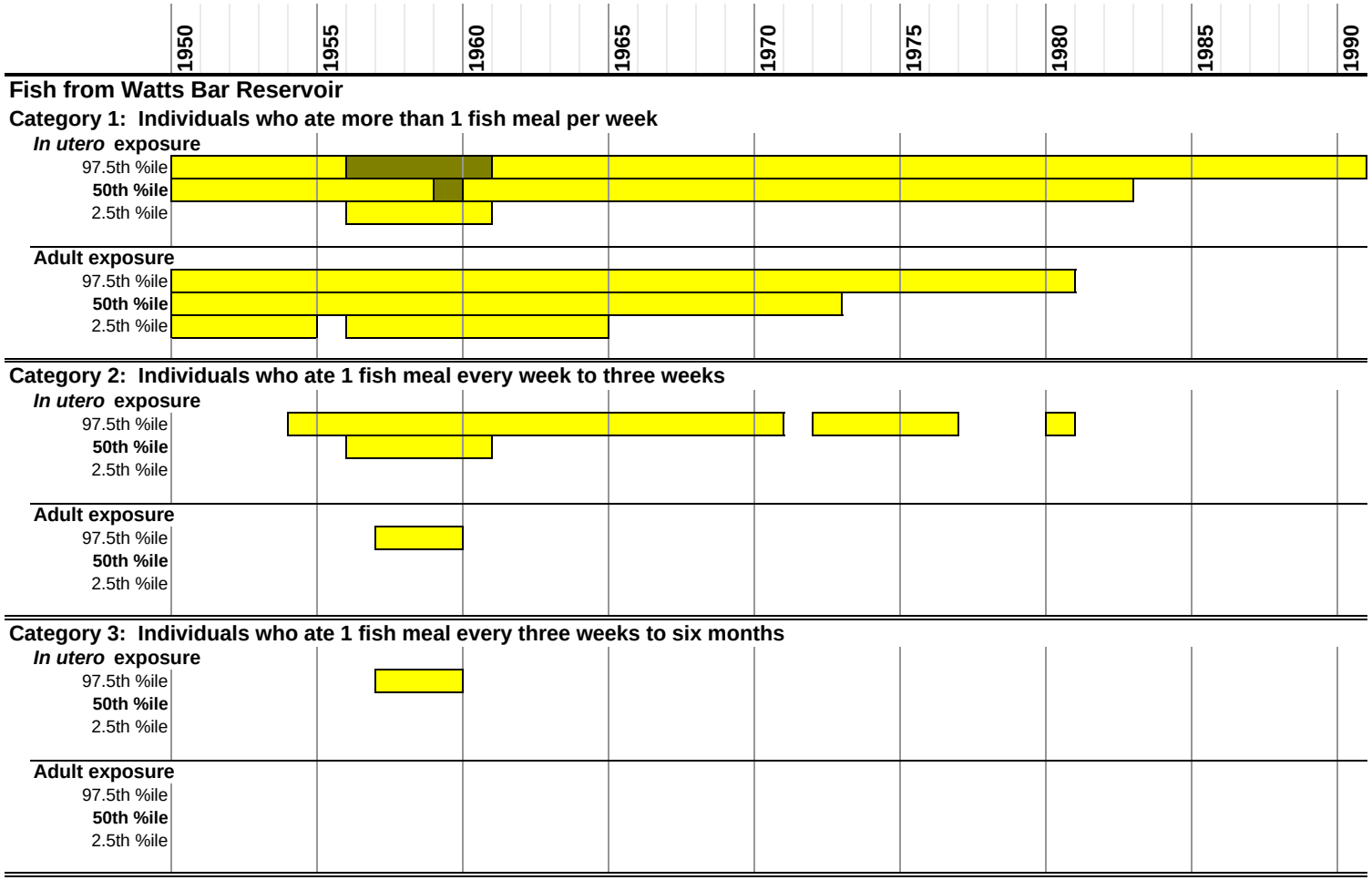
- *Comparison to RfDs and NOAELs***S** The 95% subjective confidence interval on estimated methylmercury doses from consumption of EFPC fish by members of the *Scarboro Community Residents* and *EFPC Floodplain Farm Family* populations exceeded the USEPA RfD for methylmercury (based on *in utero* exposures) for all years evaluated in this assessment (1950-1990) at the 97.5th percentile. However, doses for these populations did not exceed the NOAEL.

  Interviews with Oak Ridge area residents, including residents of the Scarboro Community and people who historically lived near EFPC, suggest that the maximum rate of consumption of fish from EFPC was about one fish meal per month. Consequently, Category 3 is the only category of fish consumer likely to have existed for EFPC. In this assessment, the average consumption rate of fish from EFPC for adults was assumed to be about 2.5 meals per year.

- *Exposures to children***S** Children who ate more than 1 meal of fish per year from EFPC may have been exposed to methylmercury at doses that exceeded the USEPA RfD. If they ate 2 or more meals of fish per year from EFPC during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

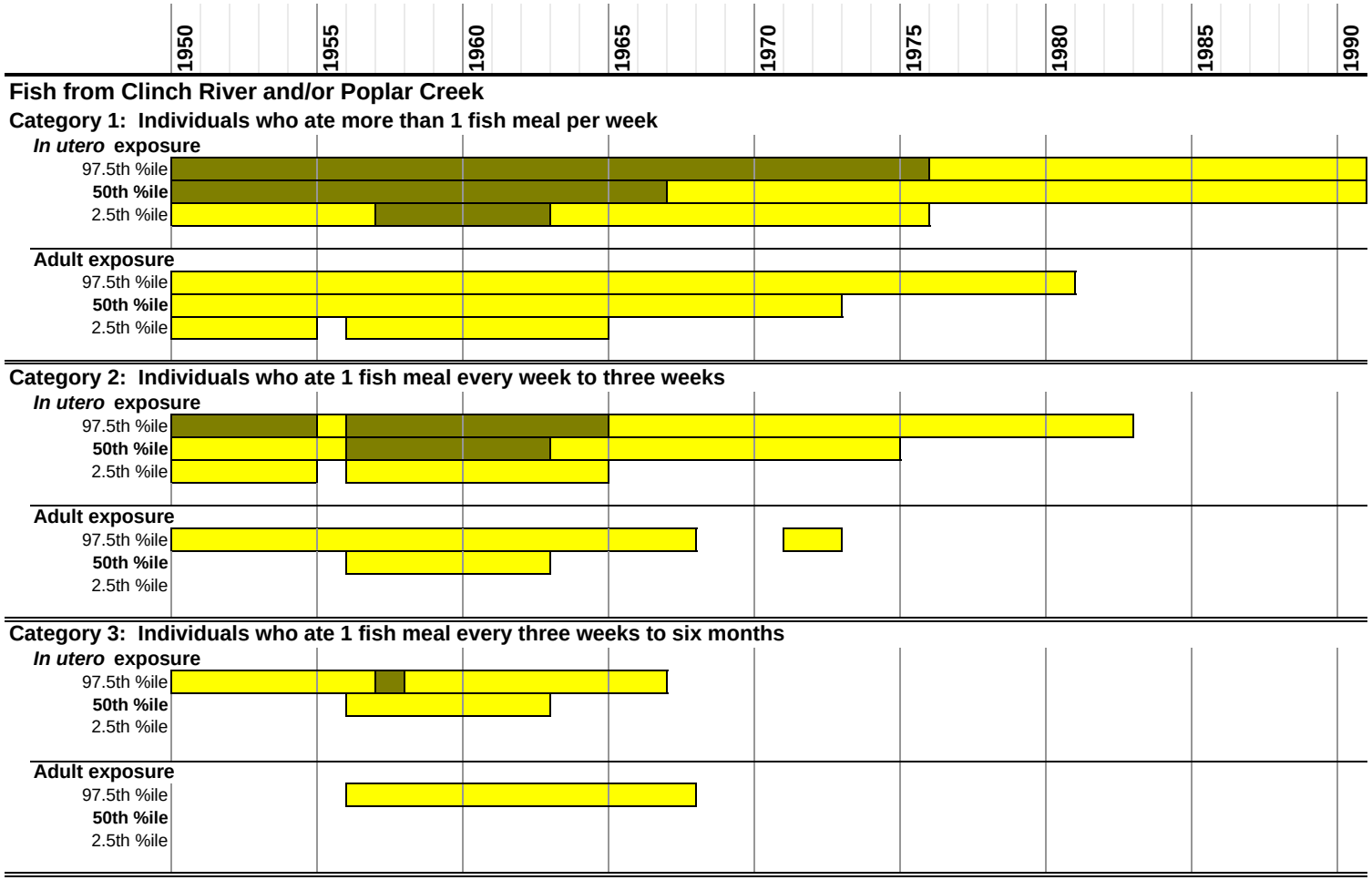
- *Exposures to adults***S** Adults who ate 2 to 3 or more meals of fish per year from EFPC may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate more than 5 meals per year during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

Figure ES-13: Years that the Estimated Mercury Doses from Consumption of Fish Exceeded the RfDs and the NOAEL-- Categories of Fish Consumers (page 1 of 3) [a]



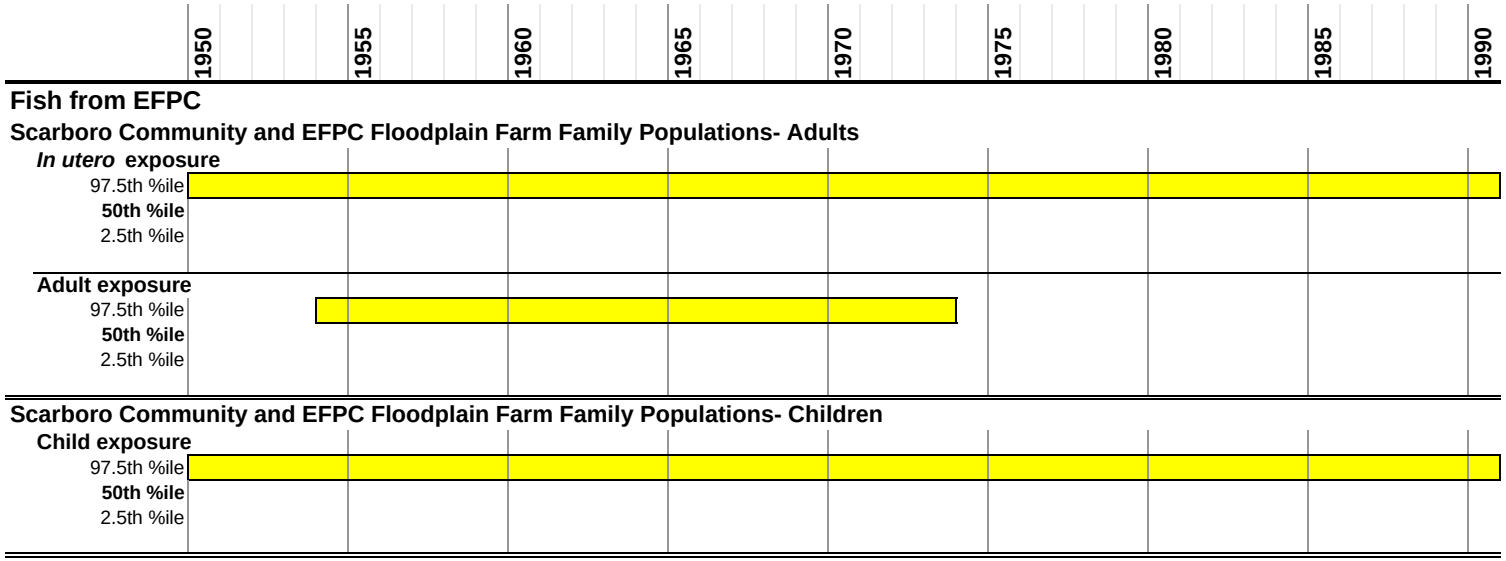
a Years that exceeded the RfD are indicated with light shading (in utero RfD = 0.0001 mg kg^{-1} d^{-1}; adult RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0005 mg kg^{-1} d^{-1})

Figure ES-13: Years that the Estimated Mercury Doses from Consumption of Fish Exceeded the RfDs and the NOAEL-- Categories of Fish Consumers (page 2 of 3) [a]



a Years that exceeded the RfD are indicated with light shading (in utero RfD = 0.0001 mg kg^{-1} d^{-1}; adult RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0005 mg kg^{-1} d^{-1})

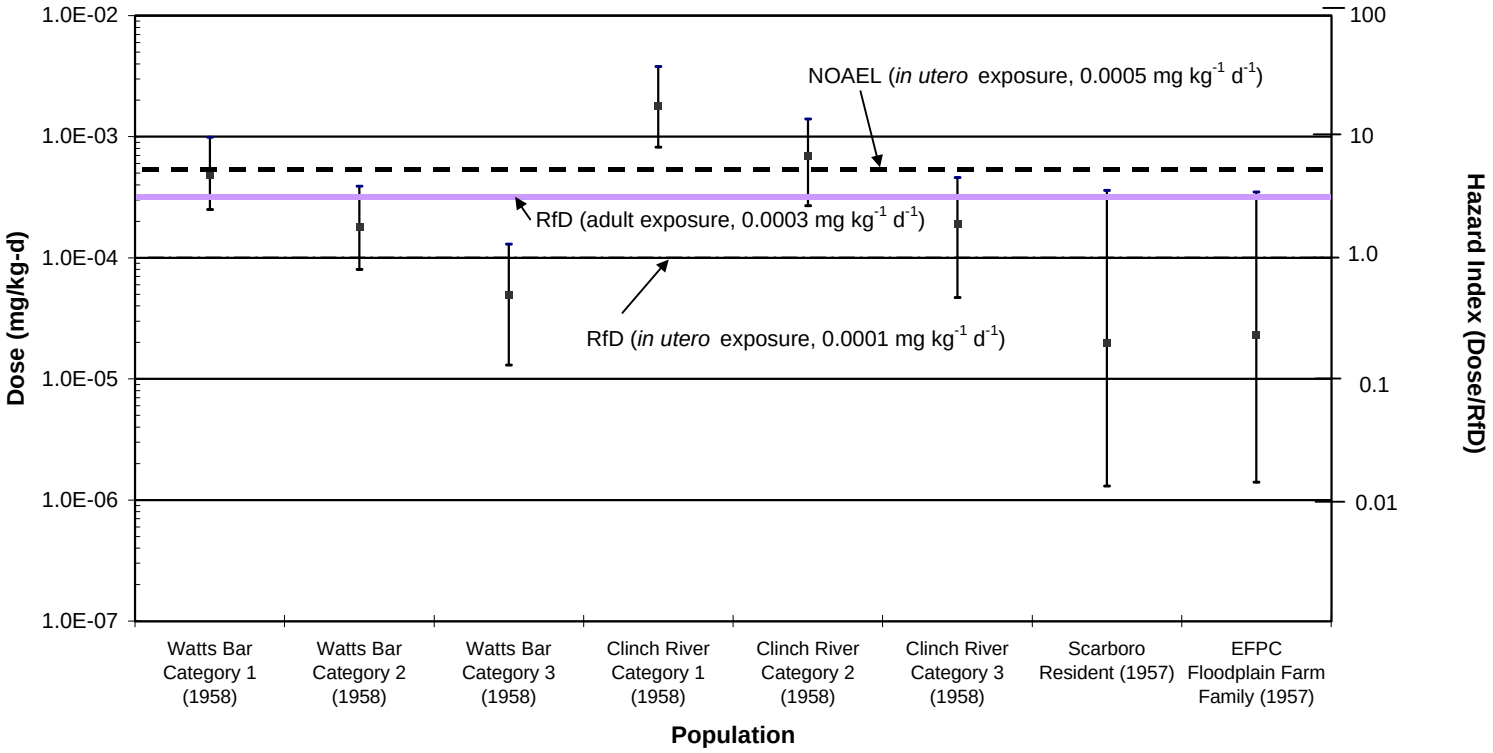
Figure ES-13: Years that the Estimated Mercury Doses from Consumption of Fish Exceeded the RfDs and the NOAEL-- Categories of Fish Consumers (page 3 of 3) [a]



a Years that exceeded the RfD are indicated with light shading (*in utero* RfD = 0.0001 mg kg^{-1} d^{-1}; adult RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0005 mg kg^{-1} d^{-1})

(None on this page)

Figure ES-14: Methylmercury (Adult exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values



Based on the results of the dose reconstruction for mercury and the comparison of estimated doses to toxicity benchmark values, the Task 2 team concluded that the following behaviors may have resulted in exposure to mercury at annual average doses above the RfDs:

Behaviors that may have resulted in exposure to mercury at annual average doses above the RfDs	
~	**Consumption of any fish** from EFPC, the Clinch River, or Poplar Creek
~	**Consumption of more than 3 or 4 meals of fish** per year from Watts Bar Reservoir
~	**Consumption of fruits or vegetables that grow above-ground from backyard gardens** in the Scarboro Community or within several hundred yards of the EFPC floodplain
~	**Playing in EFPC** more than 10-15 hours per year
~	**Living or attending school** within several hundred yards of the EFPC floodplain or in the Scarboro Community (from inhalation of airborne mercury)

The likelihood that these behaviors resulted in annual average doses above the RfDs was greatest during the period of highest mercury releases from Y-12 (that is, the mid-1950s to early-1960s).

While the results of the dose reconstruction for mercury indicate that exposures through inhalation, consumption of above-ground fruits and vegetables, contact with EFPC water and sediment, contact with EFPC floodplain soil, and consumption of fish may have resulted in annual average doses above the RfDs for mercury for some populations and some years, the results also show that annual average doses through some exposure pathways were likely insignificant, even during the years of highest mercury releases from Y-12.

Based on the results of the dose reconstruction for mercury, the Task 2 team concluded that the following behaviors were *not likely* to have resulted in exposure to mercury at annual average doses above the RfDs:

Behaviors *not likely* to have resulted in exposure to mercury at annual average doses above the RfDs	
~	**Consumption of beef** from cattle that grazed in the floodplain or downwind of Y-12
~	**Consumption of fruits or vegetables from backyard gardens** located more than one mile from the EFPC floodplain (with the exception of the Scarboro Community during the 1950s and early-1960s)
~	**Living or attending school** more than 1-mile from the EFPC floodplain (from inhalation of airborne mercury; except for in the Scarboro Community in the 1950s and early 1960s).

This page intentionally left blank.

1.0 INTRODUCTION

Between 1953 and 1962, extremely large quantities of mercury were used at the Oak Ridge Reservation's Y-12 Plant, in Oak Ridge, Tennessee, in a crash program to produce enriched lithium for use in thermonuclear weapons. Many of the details surrounding the use of mercury were top secret. Studies of the off-site environment beginning around 1970 showed high concentrations of mercury in soils, sediments, and fish downstream from the Y-12 Plant. Subsequent investigations of mercury usage at Y-12, initiated in the mid-1980s in response to public concern over the potential for adverse health effects from mercury exposure, showed that large quantities of mercury had been released from the Y-12 Plant to air and surface water, particularly during the 1950s and early 1960s.

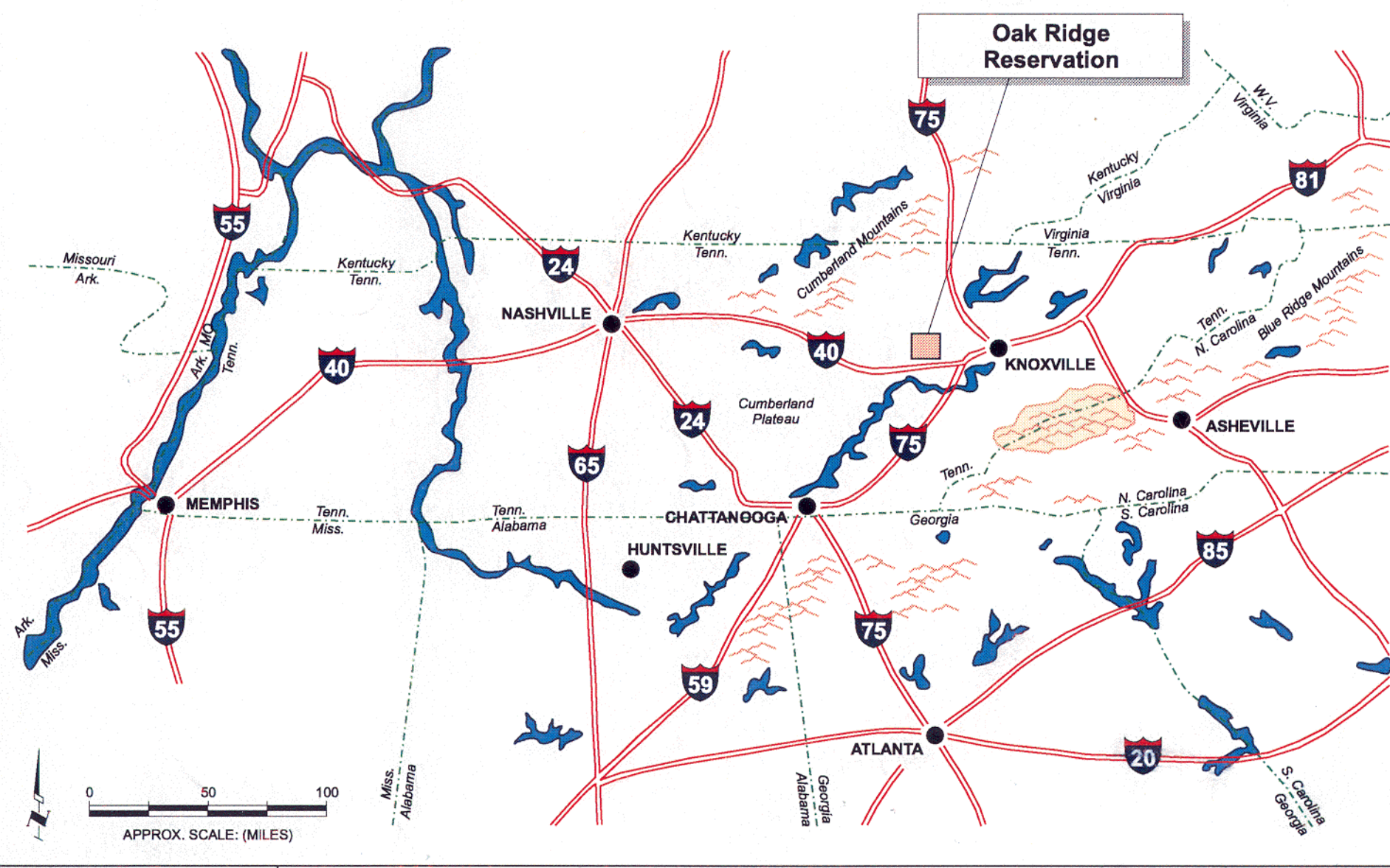
In 1991, the State of Tennessee and the United States Department of Energy (DOE) entered into the Oak Ridge Health Studies Agreement. Under the agreement, DOE agreed to provide the State of Tennessee with funds to conduct an independent assessment of human health risks to off-site populations that may exist as a result of past activities at DOE's Oak Ridge complex. Task 2 of the Oak Ridge Dose Reconstruction focused on reconstructing mercury doses to people who lived in and around Oak Ridge between 1950 and 1990, who may have been exposed to mercury released from the ORR.

1.1 Site Description

The ORR is located in eastern Tennessee (Figure 1-1), approximately 25 miles west-northwest of Knoxville, and includes parts of Anderson and Roane Counties. The following sections describe the ORR and the surrounding areas in more detail, particularly the Y-12 Plant and the areas affected by mercury releases from Y-12.

1.1.1 The Oak Ridge Reservation and the Y-12 Plant

The Army Corps of Engineers began to develop the ORR, originally known as the Clinton Engineer Works, in 1942 as one of several facilities being constructed nationwide under the top-priority, top-secret Manhattan Project. The original mission of this project was to supply special nuclear materials for the research, development, and production of the first atomic bomb. The Oak Ridge site was selected in part because of the seclusion provided by the repeating sequences of elongated ridges and intervening valleys that characterize this part of eastern Tennessee, as well as the availability of large amounts of electrical power from nearby Tennessee Valley Authority (TVA) hydroelectric facilities. Within two years, the ORR became the site of massive development efforts at three main plant areas, each with a code name to disguise its identity– K-25, X-10 (later known as Oak Ridge National Laboratory or ORNL), and Y-12 (Figure 1-2). Each plant was located in a different valley within a tightly controlled security area.

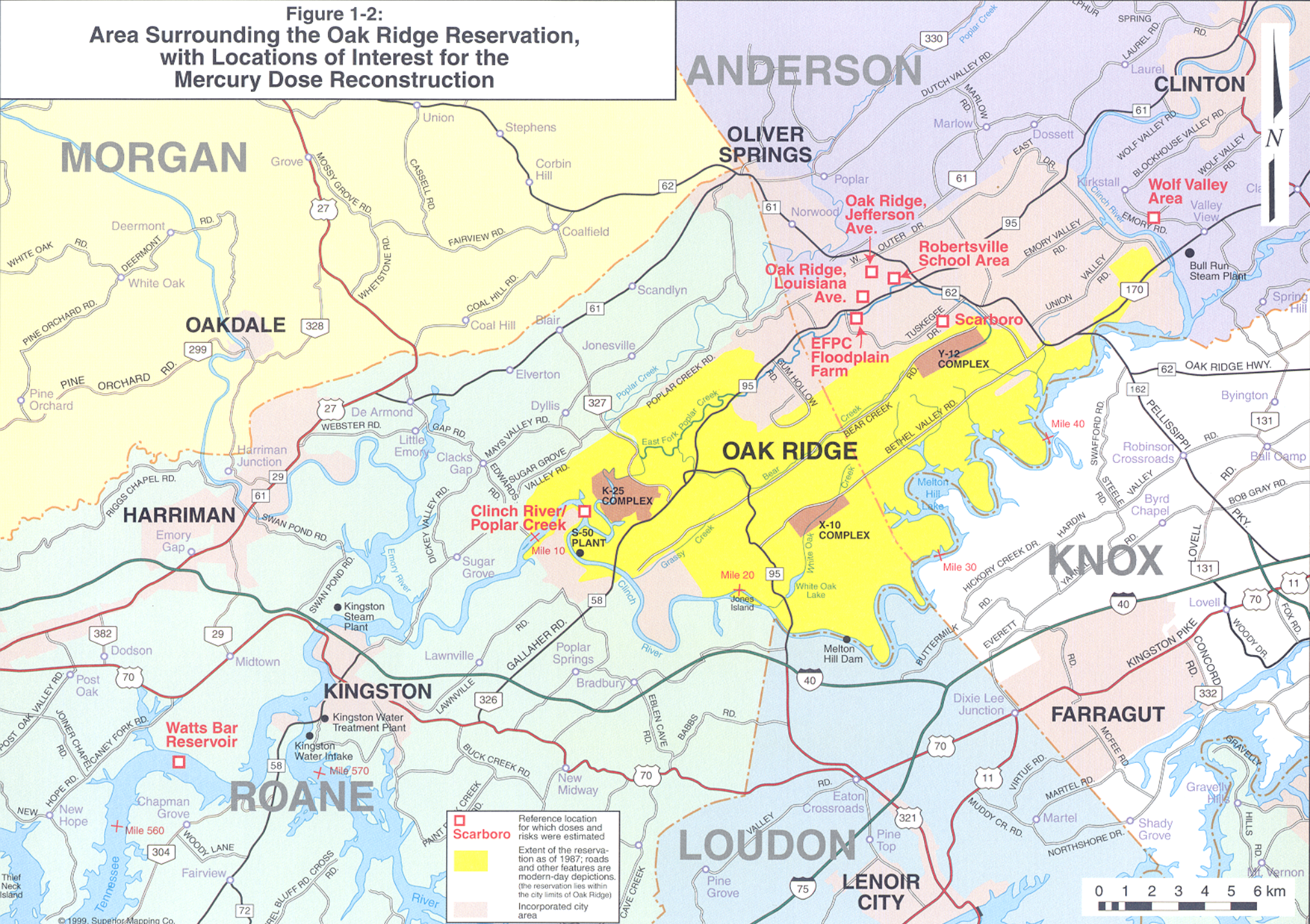


ChemRisk
A Division of McLaren/Hart

FIGURE 1-1

OAK RIDGE AND SURROUNDING AREA

Figure 1-2:
Area Surrounding the Oak Ridge Reservation, with Locations of Interest for the Mercury Dose Reconstruction



This page intentionally left blank.

The X-10 site saw development of the world's first full-scale nuclear reactor, then called the Clinton Pile, to demonstrate the production of plutonium from natural uranium fuel. Development of processes to enrich uranium in its uranium-235 isotope gave rise to the K-25, S-50, and Y-12 Plants. Multiple enrichment processes were concurrently developed because it was not clear which processes would work satisfactorily or which would be most efficient. The K-25 Plant was built to perform enrichment by the gaseous diffusion process. The S-50 Plant was built near the K-25 Plant to demonstrate the liquid thermal diffusion process**S** after the gaseous diffusion process was chosen as the method of choice for uranium enrichment, the S-50 Plant was shut down.

The Y-12 Plant is located on the eastern end of the ORR in Bear Creek Valley**S** it is bordered on the north by Pine Ridge and on the south by Chestnut Ridge. Its original, primary mission was to enrich uranium by the electromagnetic process using devices called calutrons. Following World War II, Y-12 was transformed into a high-tech plant for processing nuclear materials and production of weapons components. In the early 1950s, when the United States launched a crash program to produce enriched lithium deuteride (LiD) for use as a fuel in thermonuclear weapons (UCCND 1983a), Y-12 was given the assignment to separate high-purity lithium-6 (^{6}Li) from natural lithium to produce enriched ^{6}Li deuteride for use in more powerful thermonuclear weapons (UCCND 1983a).

Pilot scale tests conducted at Y-12 between 1950 and 1955 showed that a chemical exchange process known as Colex was the most efficient industrial-scale process for enriching lithium in ^{6}Li (USDOE 1993). In the Colex process, lithium isotopes were separated by transferring them between an aqueous (water-based) solution of lithium hydroxide and a solution of lithium in mercury. Between December 1953 and September 1955, two large-scale production facilities for enrichment of lithium were completed at Y-12. These continued to operate until 1962, when production of enriched lithium ceased (Richmond and Auerbach 1983). Like most of the subsequent missions at Y-12, this mission was highly classified, . In total, about 24 million pounds (over 1,000 yd^3 by volume) of mercury were used in the process (USDOE 1993). Most of the mercury losses to the environment from Y-12 occurred during this period (UCCND 1983a).

1.1.2 The City of Oak Ridge and Surrounding Environs

When the Army Corps of Engineers began construction on the ORR in 1942, all of the original farm families were moved from the site of construction. About 3,000 individuals received court orders to vacate their homes within weeks, and a "workers' city" was constructed on the northeastern edge of the ORR along the valley of East Fork Poplar Creek (EFPC). In 1943, employees named the city "Oak Ridge." By 1945, the population of Oak Ridge reached 75,000 persons. However, during this same year, the original mission of the ORR ended and, by 1946, over 40,000 people had left Oak Ridge. In 1949, the government-owned town was opened to visitors, and in 1950, the Scarboro Community, located in an isolated valley just north of the Y-12 Plant on the opposite side of Pine Ridge, was completed. Scarboro, established as a residential area for African American employees of the ORR facilities, is the closest community to any of the three Oak Ridge facilities, being about one-third mile north of the ORR boundary. The city of Oak Ridge was incorporated in 1959.

Although the Y-12 Plant is located within the present-day corporate limits of the city of Oak Ridge, it is separated from the city center by Pine Ridge, which largely impedes the exchange of air between Y-12 and the city (U.S. Weather Bureau 1953; Gifford 1995). Pine Ridge rises to about 300 feet above the valley floor and is heavily wooded. Chestnut Ridge, to the south of Y-12, is not as high nor as steep as Pine Ridge, but is also wooded (Bailey and Lee 1991). Meteorological data collected at a tower near the east end of the Y-12 Plant show that winds blow predominantly northeast and southwest along the valley occupied by Y-12 between Pine Ridge and Chestnut Ridge, following the local terrain.

EFPC, which originates from a spring beneath the Y-12 Plant, is initially confined to a man-made channel and flows northeasterly through the Y-12 Plant along the axis of Bear Creek Valley. At one time, EFPC received drainage from more than 200 individual Y-12 waste water outfalls (Loar et al. 1989). In 1963, New Hope Pond was constructed on EFPC at the east end of the plant to neutralize effluent from Y-12 and to serve as a settling basin for heavy metals and solids (Bailey and Lee 1991). In 1988, Lake Reality was constructed and flow from Y-12 was diverted past the New Hope Pond site through Lake Reality.

Beyond New Hope Pond/ Lake Reality, EFPC flows north through a gap in Pine Ridge and into the city of Oak Ridge. Historically, the most highly developed residential and business section of Oak Ridge was the northeast end of the EFPC watershed (TVA 1959), although several residential developments were built within the EFPC watershed west of the point where EFPC makes a sharp turn to the west. Beyond this point, EFPC flows approximately 12.5 miles to join Poplar Creek about 5.5 miles above Poplar Creek's mouth on the Clinch River. Below Poplar Creek, the Clinch River flows approximately 12 miles to join the Tennessee River at approximately Tennessee River Mile (TRM) 568. The first impoundment on the Tennessee River downstream of the Clinch River is Watts Bar Dam, located at approximately TRM 530. Watts Bar Reservoir extends 72 miles above Watts Bar Dam, and includes part of the Clinch River below its confluence with Poplar Creek.

Numerous floods have been reported on EFPC over the last 200 years. Floods occur on the creek at an average rate of approximately four per year (TVA 1959). Year-round, flow in EFPC is maintained by effluent from the Y-12 Plant, which contributes as much as 20 $ft^3 s^{-1}$ (1 $ft^3 s^{-1}$. 0.65 million gallons per day). The city of Oak Ridge municipal waste water treatment plant at approximately EFPC Mile 8 adds as much as 10 $ft^3 s^{-1}$ to the EFPC flow. Y-12 and the city of Oak Ridge contribute about half of the total flow in EFPC**S** the 40-year annual average flow of EFPC near its junction with Poplar Creek, about 14 creek miles downstream from Y-12, is approximately 65 $ft^3 s^{-1}$.

In December 1982, the Tennessee Department of Health and Environment (TDHE) posted warning signs on EFPC advising against consuming fish from EFPC because of contaminants released from Y-12, including mercury. On May 17, 1983, in response to state pressure and Freedom of Information Act inquiries by a local newspaper, DOE released information disclosing that two million pounds of Y-12's historical inventory of mercury was lost or unaccounted for, a significant part of which was released to the environment in discharges to EFPC.

1.2 Objectives of the Task 2 Dose Reconstruction

Screening analyses performed during the Dose Reconstruction Feasibility Study to identify important environmental exposure pathways and materials released from the ORR showed that mercury was potentially one of the most significant materials used at the ORR in terms of non-carcinogenic health hazards to off-site populations (ChemRisk 1993b). Task 2 of the Oak Ridge Dose Reconstruction examined potential health hazards associated with mercury releases from the ORR in greater detail.

The specific objectives of Task 2 of the Dose Reconstruction Study were to:

- describe and independently quantify past releases of mercury from the ORR;
- characterize environmental concentrations of mercury from those releases;
- define potential pathways of human exposure to mercury;
- describe potentially exposed populations;
- estimate historical human exposures and doses; and
- estimate human health hazards, to put the dose estimates in perspective.

This document summarizes the methods and results of the Task 2 Dose Reconstruction.

1.3 Document Structure

Subsequent sections of this document are organized as follows:

Section 2.0, Overall Approach– presents an overview of the approach used by the project team to characterize historical releases of mercury from the ORR and reconstruct potential doses to off-site populations, including characterization of uncertainties in these dose estimates.

Section 3.0, Historical Mercury Operations on the ORR– summarizes historical uses of mercury on the ORR and events leading to formation of the Mercury Task Force in 1983.

Section 4.0, Source Term Assessment– summarizes methods and results of the project team's independent evaluation of quantities of mercury released to the environment.

Section 5.0, Measurements of Mercury in the Environment Near the ORR– describes historical environmental monitoring programs conducted by ORR workers and other groups to characterize mercury concentrations in the environment near the ORR, and summarizes results of environmental measurements. In addition, this section describes studies to identify the chemical and physical forms (species) of mercury in the environment, and discusses the species of mercury assumed to be present in each medium.

Section 6.0, Identification and Characterization of Exposure Pathways and Potentially Exposed Populations– describes exposure pathways and off-site populations evaluated in the dose reconstruction for mercury.

Section 7.0, Estimation of Exposure Point Concentrations in Surface Water, Air, Soil, and Fish– describes exposure point concentrations estimated by the project team for each medium, based on concentrations of mercury measured in the off-site environment or calculated from estimates of historical releases.

Section 8.0, Characterization of Transfer of Mercury to Vegetation and to Milk and Meat– describes factors to characterize the transfer of mercury from air and soil to vegetation and from cattle intake to milk and meat.

Section 9.0, Identification of Parameter Distributions to Characterize Exposure in Humans– describes contaminant- and population- specific exposure parameters used in this study to characterize uptake of mercury by exposed populations.

Section 10.0, Estimation of Doses to Potentially Exposed Populations– presents the results of the Task 2 dose calculations.

Section 11.0, Toxicity Benchmarks for Comparison to Estimated Mercury Doses– summarizes published literature and ongoing investigations of adverse health effects at various levels of mercury exposure at locations throughout the world.

Section 12.0, Estimation of the Potential for Health Effects in Exposed Populations– combines the dose reconstruction results and mercury toxicity information to estimate the potential for adverse health effects from exposure to mercury released from the ORR.

Section 13.0, References– provides the references used in this assessment.

2.0 OVERALL APPROACH

Mercury doses to off-site populations who lived near Y-12 between 1950 and 1990 were estimated based on historical release information, historical environmental measurements, and assumptions about where people lived and their activity patterns. The following sections describe the general approach used to estimate exposures in this dose reconstruction.

2.1 Estimating Dose

The term "dose" describes the amount of a substance taken in by an individual over a period of time from a variety of sources, including soil, water, food, or air, by such exposure routes or "pathways" as ingestion, inhalation, or absorption through the skin. Health hazards resulting from exposure are generally correlated with the magnitude of the dose. The goal of the current evaluation is to estimate the likely range of historical mercury doses to individuals for several populations who historically lived near the Y-12 Plant.

One approach to estimating an individual's dose following exposure to a chemical agent is to directly measure how much of the chemical gets into the body. However, direct measurements of uptake following environmental exposures are generally not available, particularly when the exposures being evaluated occurred in the past or individuals are not aware that the exposures are occurring. Alternatively, the dose can be estimated using mathematical models or equations that take into account:

- The concentration of the chemical in a contaminated medium (such as soil, water, food, or air),
- The volume of the contaminated medium that an individual contacts, ingests, or inhales, and
- The amount of the chemical in the contacted, ingested, or inhaled medium that is actually absorbed into the blood stream.

Typically, information describing the volume of soil, water, food, or air that an individual contacts, ingests, or inhales is presented as a daily rate (for example, grams of soil ingested per day, liters of water ingested per day, kilograms of food ingested per day, or cubic meters of air inhaled per day), averaged over the length of the individual's exposure period (for example, one week, one year, or multiple years). Estimates of an individual's daily dose of a contaminant are typically normalized to the person's body weight.

An individual's average daily dose of a chemical over an exposure period can thus be estimated using the following mathematical formula:

$$Dose = \frac{C \times U \times EF \times ED}{BW \times AT} \quad (2.1)$$

where:

Dose = Average amount of the chemical taken in by an individual per day from a single exposure pathway, per kilogram of body weight (mg kg^{-1} d^{-1});

C = Concentration of the chemical in the contacted exposure medium (for example, mg m^{-3} in air, mg L^{-1} in water, or mg kg^{-1} in soil);

U = Daily intake rate of the exposure medium [for example, breathing rate of air (m^3 d^{-1}), ingestion rate of food (kg d^{-1}), drinking rate of water (L d^{-1}), or dermal contact rate with soil (mg cm^{-2} d^{-1})];

EF = Exposure frequency [number of days per exposure period that exposure occurred (for example, d wk^{-1}, d y^{-1})];

ED = Exposure duration (for example, 1 week or 1 year);

BW = Body weight of the exposed individual (kg); and

AT = Averaging time (period of time over which exposure is averaged, equivalent to the exposure duration expressed as the number of days exposed).

Because the current assessment focuses on reconstructing doses that occurred in the past, concentrations of mercury contacted by nearby residents must, for the most part, be estimated based on historical information that describes how mercury was used at Y-12 and how much of the mercury was released to water or air during different time periods. An extensive part of the current investigation focused on careful review of classified and unclassified information on mercury use and releases from Y-12 and historical environmental monitoring studies, as well as independent estimation of how much of the mercury was released to air and surface water (described in detail in Sections 3, 4, and 5).

Mercury is found in the environment in many different forms or species. Typically, these species are grouped in three general forms: elemental mercury, inorganic mercury, and methylmercury. Each form, at sufficiently high exposure concentrations, has been associated with different noncarcinogenic, or systematic, health effects, and doses for each of these three forms are typically estimated separately. For example, elemental mercury has been associated with neurological effects in workers who were repeatedly exposed to very high air concentrations. Inorganic mercury has been associated with kidney effects in laboratory

animals who were administered very high doses in their food or water. And methylmercury has been associated with neurological effects in children whose mothers ingested high concentrations of methylmercury in contaminated seed grain during their pregnancies. None of the three forms of mercury has been conclusively shown to be carcinogenic . In the current assessment, elemental mercury, inorganic mercury, and methylmercury doses are estimated separately. Assumptions about the species of mercury assumed to be present in each medium are described in Section 5.

Values for each of the parameters in the above dose equation will differ depending on the individuals being evaluated. For example, a child who lives near a creek is likely to contact surface water or sediment in the creek more frequently than a child who lives some distance away. Similarly, an individual residing near a release source may be exposed to a higher concentration of a contaminant than an individual living further away. In this assessment, historical mercury doses were estimated for several populations who lived downwind or downstream of Y-12 and may have been exposed to mercury released from Y-12. These populations were:

- *Residents of the Wolf Valley area*, who lived approximately five miles down Union Valley from Y-12, in the direction of predominant wind flow. These individuals may have been exposed to mercury in direct airborne releases from Y-12.

- *Residents of the Scarboro Community*, who lived approximately one-third mile north of the Y-12 boundary on the opposite side of Pine Ridge. These individuals may have been exposed to mercury in direct airborne releases from Y-12 and mercury that volatilized to air from EFPC. In addition, members of this community visited EFPC for recreational activities, and may have come in contact with mercury in contaminated surface water and sediment, or may have occasionally caught and consumed fish from the creek.

- *Students at Robertsville School*, located along the banks of EFPC at approximately EFPC Mile 12. These junior high school students may have been exposed to mercury that volatilized to air from EFPC, or may have contacted mercury in contaminated surface water, sediment, or floodplain soil.

- *Members of families who lived on farms along EFPC.* These individuals may have been exposed to mercury that volatilized to air from EFPC. In addition, some of these families grew vegetables in backyard gardens and raised beef and dairy cattle that may have resulted in exposure to mercury in contaminated vegetables, meat, or milk. Members of these families also occasionally recreated in EFPC and may have contacted mercury in contaminated surface water and sediment, or may have occasionally caught and consumed fish from the creek.

- *Residents of single- and multi-family homes just outside of the EFPC floodplain.* These individuals may have been exposed to mercury that volatilized to air from EFPC.

- *Anglers who caught and consumed fish from downstream waterways*, including Poplar Creek, the Clinch River, and Watts Bar Reservoir. These individuals may have been exposed to mercury that concentrated in the fish.

These populations are described in greater detail in Section 6.

In this assessment, estimated daily doses for different population groups were averaged over an assumed exposure duration of one year, and the exposure duration and averaging time were assumed to be equal (and thus, the *exposure duration* and *averaging time* parameters canceled out in the dose equation). Exposure concentrations and exposure parameters used in this assessment for different exposure populations and different points in time are described in greater detail in Sections 7, 8, and 9.

For most noncarcinogens, the likelihood of adverse health effects from lower dose exposures over a period of time is estimated based on the average daily intake of the chemical over that time period (in this assessment, one year), normalized to the individual's body weight. To provide some perspective on the likelihood that these doses may have resulted in an adverse health effect, these normalized average daily doses are then compared to a threshold dose based on exposures averaged over a similar period of time. Typically, it is assumed that if the normalized average daily dose is below the threshold dose for a given health effect, the health effect will not occur. Because thresholds may vary for different individuals within a population, and because thresholds are often based on information from limited studies (often in laboratory animals), regulatory agencies have developed recommended doses of concern for noncarcinogens that include "safety factors." For example, the US Environmental Protection Agency (USEPA) has established Reference Doses (RfDs) for each of the three forms of mercury. RfDs represent normalized averaged daily doses at which no adverse health effects are expected, and are based on threshold doses combined with safety factors (which typically range from 10 to 1,000) to account for uncertainties in extrapolating from threshold doses to actual exposure situations in human populations. In this assessment, normalized average daily doses are compared to both threshold doses and agency-recommended RfDs (described in detail in Section 11).

Generally, most published threshold doses and most established RfDs or other regulatory criteria for environmental contaminants are based on the dose of a chemical *administered* to an animal in food or water or to which a human is exposed in an occupational setting. These regulatory criteria or threshold doses are *not* generally based on the *absorbed* dose. However, doses to which an individual is exposed in an environmental setting may be absorbed into the body at a rate that is greater or lesser than the rate associated with the conditions under which the regulatory criteria or threshold doses were established, even if the external exposing masses were the same. Failure to adjust for differences in uptake may result in

over- or under-estimates of risks associated with a given dose. In this assessment, differences in the rate of uptake between the two scenarios are accounted for by multiplying the calculated dose (Equation 2.1) by the *relative bioavailability (B)* of the chemical. Derivation of relative bioavailability factors for different species of mercury is discussed in Sections 5 and 9.

2.2 Information Sources for the Mercury Dose Reconstruction

Information used in the Task 2 dose reconstruction for mercury released from Y-12 was obtained from a number of sources, including:

- Published and unpublished information on Y-12 Plant history and releases;
- Historical environmental monitoring data for mercury near the ORR;
- Interviews with current and former ORR employees;
- Published information on the environmental fate and toxicity of mercury;
- Published information on rates of intake of food, air, water, and other exposure media by individuals living in the rural southeastern United States during the 1950s and 1960s; and
- Consultation with experts on the behavior and toxicity of mercury, including scientists at Oak Ridge National Laboratory (ORNL) and the Agency for Toxic Substances and Disease Registry (ATSDR), and contacts gained through several technical meetings, including the USEPA Workshop on Mercury Speciation (Denver, CO, September 1996).

2.3 Characterizing the Uncertainty and Variability in Dose Estimates

The majority of inputs to the dose equations used to estimate historical mercury doses are not known perfectly. In many cases, there is a lack of knowledge about the true value of a parameter. For example, in the process of dose reconstruction for releases of mercury from the ORR, there are incomplete records and data about historical operations and environmental conditions leading to human exposure, resulting in uncertainties in release estimates and model predictions. In addition, values for many of the parameters in the dose equation may vary between individuals (for example, body weights and rates of food and water ingestion). Some parameters reflect both informational uncertainty and interindividual variability.

To estimate the range of doses likely for a population of interest and the uncertainty about the doses to individuals within the population, each parameter in the dose equations can be represented by a range of values that describes what is known about the uncertainty and variability in the value of that parameter for that population. Probability analysis is a process that allows computation of the total uncertainty and

variability induced in the output by quantified uncertainty and variability in the inputs and models. In probability analysis, most inputs to the dose calculations are quantified not in terms of a single, discrete number, but as probability density functions (PDFs). These PDFs are similar to frequency distributions and quantitatively express the existing knowledge about alternative values for a parameter.

It is likely that some values within each range of parameter values will occur more frequently within a given population than other values. For example, one or two individuals within a population may have an unusually low body weight and one or two individuals may have an unusually high body weight, but most individuals will have a body weight somewhere in the middle. To develop an accurate estimate of the possible range of normalized average daily doses within a population, distributions are established such that parameter values that are more likely in a population will be selected more frequently. To accomplish this, PDFs are typically described as normal, lognormal, discrete, uniform, or triangular distributions. In the current assessment, PDFs were established based on a number of sources of information, including site-specific data, professional judgement after review of the literature, and consultation with experts.

When inputs to a dose equation are defined by distributions, each equation has many possible answers and must be solved repeatedly using different values selected from the distributions of input parameter values. In the current assessment, this process was accomplished using a method known as Monte Carlo simulation. In Monte Carlo simulation, an outcome (such as an estimate of dose) is calculated repeatedly using, in each trial, values selected from the PDF for each uncertain parameter. Selected values are more likely to be drawn from the areas of the PDF that have higher probabilities of occurrence. The result of the simulation is itself a PDF, describing not only the best estimate of the overall result, but also the uncertainty in the overall result that is induced by the uncertainty in the input parameters. While the simulation process is very complex, commercial computer software programs perform the calculations as a single operation. In the current assessment, Monte Carlo simulations were conducted using the Crystal Ball Pro (version 4.0) software package[1]. The inputs and results of the probability analysis for the current assessment are presented in this report in the following manner:

- Throughout the current assessment, efforts were made to identify and quantify the uncertainty and variability in input parameters.

- For each uncertain parameter presented in this report, a PDF was defined as a subjective confidence interval within which there is a high probability of encompassing the true but unknown parameter value for a given population or situation. Whenever possible, site- or region-specific information was used to develop PDFs. In addition, an effort was made to base the PDFs on the specific time periods of interest (1950s to present).

[1]Crystal Ball® Version 4.0, Decisioneering, Inc. Aurora, CO. 1996.

- For each dose equation describing exposure to a specific population via a specific pathway, a Monte Carlo simulation was run, using a sample size of 500 trials, for each year of interest.

- Results from the Monte Carlo simulations are presented as tables and graphs describing the 2.5, 50, and 97.5% confidence levels of a PDF representing the dose estimated for each pathway.

The resulting distributions of dose are intended to reflect the population risk, and include the central tendency and low- and high-end portions of the risk distribution. These distributions of dose are intended to reflect exposures to average individuals within the population, as well as more highly and less exposed individuals.

This page intentionally left blank.

3.0 HISTORICAL MERCURY OPERATIONS ON THE ORR

The following section describes historical operations at the ORR involving mercury use, and past investigations into the quantities of mercury used, recovered, and lost to the environment. The information summarized here was gathered from review of historical literature and interviews with current and former ORR staff. In particular, this section describes:

- **Lithium isotope separation operations at Y-12 in the 1950s and 1960s**— these were the ORR operations that used the largest quantities of mercury, and included six pilot plants and three production facilities that used over 20 million pounds of mercury, as well as several auxiliary operations that supported the lithium isotope separation process.

- **The Mercury Task Force investigation of mercury use at Y-12**— the Mercury Task Force was convened in 1983 to investigate quantities of mercury used, recovered, and lost to the environment from Y-12. The Mercury Task Force Report (UCCND 1983a) summarizes the results of their investigation.

- **Procedures historically used to monitor mercury in building air and in liquid effluents from the Y-12 Plant**— beginning in the early 1950s, monitoring programs were conducted to measure mercury in air and liquid effluents from lithium separations processes.

The results of the Task 2 effort to quantify mercury releases from lithium isotope separations and auxiliary operations at Y-12 (the mercury "source terms" used to reconstruct historical off-site doses) are described in Section 4.0. In addition to the lithium isotope separation operations, mercury was used in minor quantities in several other operations at the Y-12, X-10, and K-25 complexes. For each of these minor operations, the project team either found no evidence of mercury release or found releases were insignificant (i.e., they were less than 1 percent of the releases from the Y-12 operations described in this section). No source terms were estimated for these minor uses of mercury. Descriptions of these operations are presented in Appendix A to this report, along with information describing their significance relative to releases from lithium isotope separation operations.

3.1 Lithium Isotope Separation

Beginning in the late 1940s, United States defense program needs made the development of methods to separate lithium isotopes a national priority. As a result, Y-12 became a center of lithium isotope separation process development and operation.

Early Requirements for Lithium Isotope Separation

In the late 1940s and early 1950s, several processes being developed under the auspices of the national defense program required the separation of lithium isotopes. Natural lithium consists primarily of two isotopes– lithium-6 (^{6}Li, about 7%) and lithium-7 (^{7}Li, about 93%). These two isotopes differ greatly in their ability to capture neutrons. Because of its high neutron capture cross section, ^{6}Li readily captures neutrons to produce energetic tritium and helium ions. In contrast, the low neutron capture cross section of ^{7}Li made it a desirable material for use in atomic piles (reactors) and, in 1948, it was believed that almost 100% pure ^{7}Li could be used as a reactor coolant or heat transfer medium, especially in Aircraft Nuclear Propulsion (ANP) reactors. To provide ^{7}Li for this application, the ORNL Materials Chemistry Division initiated work in 1949 to find a method to separate lithium isotopes.

In addition, Los Alamos scientists indicated a need (not disclosed at the time) for highly enriched (30-95%) ^{6}Li. Eventually, it was revealed that the enriched lithium was to be used in the development of thermonuclear weapons (ADP History 1948-51). Thermonuclear weapons, also referred to as hydrogen bombs, derive most of their energy from the fusion, or combination, of heavy hydrogen atoms into heavier atoms. In contrast, earlier fission weapons (atomic bombs) derived their energy from the splitting of uranium or plutonium atoms. Lithium deuteride was desired for use in fusion bombs because it had the required density and machinability (UCCND 1983a). Following the Los Alamos request, 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 Oak Ridge Y-12 Plant was given the assignment to develop, design, construct, and operate a production process to produce enriched ^{6}Li. In 1953, the need for enriched lithium deuteride became especially urgent after the experimental detonation of a hydrogen bomb in Russia.

Processes with the greatest potential for producing ^{6}Li efficiently used mercury as a major component of the separation process because, under certain conditions, ^{6}Li dissolves more readily in mercury than ^{7}Li. If a solution of lithium dissolved in mercury (known as lithium amalgam) is allowed to flow in contact with a fluid containing another lithium compound, ^{6}Li atoms migrate to the amalgam and^{7} Li atoms migrate to the lithium compound in the fluid (UCCND 1983a). The use of lithium amalgam for separation of lithium isotopes was demonstrated by Lewis and McDonald in 1936. According to a presentation titled "Separation of Lithium-6 and Lithium-7 by Union Carbide Nuclear Company (3-25-57)", several hundred systems to separate lithium isotopes using organic/organic or organic/aqueous systems in place of mercury and water were investigated in order to avoid the use of large quantities of mercury, but no other systems of practical significance were found. Of the chemical methods investigated, only lithium amalgam with lithium hydroxide solution or lithium amalgam with lithium compounds dissolved in organic solvents, appeared to be practical systems for use in separation of lithium (UCNC 1957).

Starting in 1950, mercury was used as a major component in the chemical exchange process employed at the ORR for separation of lithium isotopes. The desired product of lithium separation operations was lithium deuteride containing more than the 7.5% of the ^{6}Li isotope found in natural lithium. After isotopic enrichment, enriched lithium was transferred from the amalgam phase to an aqueous phase and converted to lithium hydroxide (UCCND 1983a). The lithium was then converted from the hydroxide to the chloride,

to the metal, and finally to the deuteride. Pulverized deuteride was shaped by isostatic pressing, machined, canned in stainless steel, and assembled into thermonuclear weapons components (UCNC 1957).

Development of Lithium Isotope Separation Operations at the ORR

Three processes were simultaneously evaluated at the ORR for lithium isotope separation, in an attempt to find a process that could efficiently produce ^{6}Li. These processes were Orex, Elex, and Colex- the "ex" standing for exchange. The Orex (or organic exchange) process was designed to use an organic solvent (such as ethylene diamine or propylene diamine) instead of water as the fluid to carry the lithium, because it was believed that reaction of the water phase with the amalgam could be eliminated only by use of an organic solvent instead of a water-based solvent. Two types of Orex reflux processes were tested for production– dual temperature reflux and chemical reflux. Orex was operated on a pilot scale at Y-12 in Buildings 9733-1 and 9202 from 1951-54, and at X-10 in Building 4501 from 1953-54. However, even with an organic solvent, it was difficult to completely eliminate moisture in the system (Clewett 1953) and the Orex system proved to be less efficient than an alternative approach known as the Elex (or electrical exchange) process (ADP History 1948-51). The unresolved technical problems led to the abandonment of Orex at Y-12 in March 1954, and at X-10 in July 1954.

The most productive fluid used in the isotope separation process was identified as lithium hydroxide dissolved in water. Lithium amalgam remains in a stable state in contact with an aqueous solution only if an electric current is applied to the mixture. If the current is removed, the amalgam decomposes and the lithium reacts with the water (UCCND 1983a). The Elex process, developed and patented by Union Carbide (UCNC 1957), used mechanically driven agitators to provide contact between the amalgam phase and lithium hydroxide dissolved in water. A counter balancing electromotive force (EMF) was used to prevent amalgam decomposition. At Y-12, Elex was operated on a pilot scale in Buildings 9733-2 and 9201-2 from 1950-51, and as a production scale facility in Building 9204-4 from August 18, 1953 to March 16, 1956.

While the Elex production plant was under construction in 1953, it was realized that if amalgam decomposition could be controlled without a back EMF, and if difficulties in making and pumping amalgam could be overcome, then more conventional industrial methods could be used to separate lithium. This would substantially lower the production costs (UCNC 1957). It was observed that if the Elex electrodes were removed, the elimination of oxygen gas imparted stability to the amalgam to allow contact between the two phases without serious amalgam decomposition. This observation led to the concept of the Colex (or column-based exchange) process (Clewett 1953). The Colex process was an improvement on the Elex process, since packed columns providing more surface area for exchange were used as the contact device. Pilot scale tests indicated that the Colex process would be the most efficient industrial-scale process for enriching lithium in ^{6}Li (USDOE 1993). At Y-12, Colex was operated on a pilot scale in Building 9201-2 from 1952-55, and as a production scale facility in Building 9201-4 from 1955-62 and in Building 9201-5 from 1955-59. The process required millions of pounds of mercury. Most of the mercury losses to the environment from Y-12 occurred in the eight-year period of the Colex production scale operations from 1955 to 1962 (UCCND 1983a).

Table 3-1 presents a summary of the lithium isotope separation and auxiliary operations at Y-12 for which source terms were estimated in the current assessment. A detailed description of these mercury operations, including estimated inventories and releases, is presented in Section 4.

Table 3-1: Summary of Lithium Isotope Separation and Auxiliary Operations at Y-12 for Which Source Terms Were Estimated

Mercury Operation	Building(s) Used	Time Period of Operation
Elex Pilot Plant	9733-2 and 9201-2	1950-51
Orex Pilot Plant	9202	1953-54
Colex Pilot Plant	9201-2	1952-55
Elex Production Plant	9204-4	1953-56
Colex Production Facilities	9201-4 and 9201-5	1955-62
Mercury Recovery Facility	81-10	1957-62
Steam Plants	9401-1, 9401-2, 9401-3	1943-present

A small quantity of mercury (300 pounds) was used in a mercury-thallium alloy that was used in the production of several weapons components at Y-12 in the 1980s (Radle 1996; Ford 1983). Mercury was also used in instrumentation associated with the Y-12 uranium enrichment calutrons between 1943 and 1946 (Smith 1944), similar to the use of mercury in gaseous diffusion instrumentation at K-25.

3.2 Other Mercury Operations on the ORR

Operations using mercury at X-10 and K-25 are described in Appendix A of this report as well as Taylor (1989), LaGrone (1983),and ChemRisk (1993a). Operations at X-10 that used mercury included a small Orex pilot plant in Building 4501 that operated from 1953-54. In addition, Building 4505 provided development support for the Metallex process, which used sodium amalgam (sodium in mercury) in a process to purify thorium metal. Blanco et al. (1956) describe an experimental process called Hermex in which uranium metal was purified by dissolving it in boiling mercury. Though conducted at X-10, no indication of the building where Hermex was conducted is indicated. Mercury was reportedly spilled in several other X-10 buildings (e.g., 3592, 3503) during cleaning of Orex and Metallex equipment (Taylor 1989), and LaGrone (1983) and USDOE (1989) indicate that a small quantity of mercury may have been used in X-10 Building 3503 in fuel reprocessing research. Operations at K-25 that used mercury included distillation operations in three buildings (K-1303, K-1024 and K-1420) at various times between 1948 and the early 1980s, to clean mercury used in instrumentation (LaGrone 1983).

3.3 Mercury in the Off-Site Environment and the 1983 Mercury Task Force

The possibility that mercury releases from Y-12 to surface water (East Fork Poplar Creek, EFPC) during lithium separation processes could have reached the public through consumption of mercury-contaminated fish first became a concern in 1970. Over the next 13 years, several investigations of mercury in fish and sediments downstream of Y-12 were conducted, leading to the convening of the Mercury Task Force in 1983. Prior to public release of information in 1983, however, the results of the off-site investigations for mercury were not publically known. Events leading to the convening of the Mercury Task Force in 1983, and the development of the Mercury Task Force Report, are described below.

Measurements of Mercury in the Off-Site Environment

Between 1955 and 1961, elevated mercury concentrations in surface water were measured by the K-25 Technical Division in EFPC near its confluence with Poplar Creek and at locations further downstream in Poplar Creek and the Clinch River (Kwasnoski and Whitson 1955-1961). A 1955 internal memo from the K-25 Technical Division notes that the level of mercury in Poplar Creek at the mouth of EFPC had risen sharply (up to 1.8 mg L^{-1}) above the normal levels expected, and that Y-12 was the suspected source of the mercury (Hill 1955). Mercury was also measured in the Clinch River at a location just upstream from the K-25 Sanitary Water Supply. Members of the K-25 Industrial Hygiene Section and Medical Department indicated that clinical checks of employees showed no indication of mercury excretion or other body effect (Henry 1955).

Scientists did not become aware of the potential for mercury to be methylated by microorganisms in surface water and/or sediment, and be bioconcentrated in fish, until the late 1960s. The first reported widespread occurrence of neurological disorders associated with the ingestion of methylmercury contaminated fish occurred in the Minimata area of Japan in 1968 (ATSDR 1997). In 1970, elevated mercury levels were measured in fish in EFPC by Y-12 staff (Sanders 1970). In 1974, USDOE Oak Ridge Operations (ORO) personnel measured elevated mercury levels in EFPC sediments (Reece 1974). However, staff at other ORR facilities and the public were not aware of mercury releases from Y-12 because, at the time, Y-12 lithium enrichment operations were classified.

Between 1974 and 1975, X-10 scientists began studying Poplar Creek and EFPC to see if concentrations of contaminants warranted preparation of an environmental impact statement for the ORR. A draft report was submitted to USDOE ORO, along with a proposed monitoring program and a request for funds (Richmond and Auerbach 1983). This request was later denied (Marshall 1983). In 1975, K-25 staff began sampling Poplar Creek sediments for metals because of concerns that elevated mercury levels in lower Poplar Creek came from K-25. The ORNL draft report and the K-25 sampling program resulted in the 1976-77 study "Mercury Contamination in Poplar Creek and the Clinch River" (Elwood 1977), which showed significant mercury contamination in fish. Although it did not identify the source of mercury in Poplar Creek, the author suggested that EFPC (and consequently the Y-12 Plant) was a likely source (Elwood 1977).

The 1977 Mercury Inventory Report

In 1977, USDOE ORO asked Union Carbide to reconstruct the historical inventory of mercury at Y-12. In response, two employees spent two weeks gathering information from documents and employee interviews. The resulting 10-page report, *Mercury Inventory at Y-12 Plant 1950 through 1977* (Case 1977), indicated that about 550,000 pounds of mercury had been spilled or lost to the environment, and about 1.9 million pounds of mercury remained unaccounted for. The report was classified because the quantity of mercury used in lithium enrichment was classified at the time (LaGrone 1983).

Public Awareness of Mercury Releases

On December 5, 1981, two brothers, one an employee at ORNL and the other a United States Geological Survey (USGS) employee, collected vegetation at Y-12 near EFPC. They were seeking data to justify a joint ORNL-USGS research project. The ORNL employee had become aware of elevated mercury levels in EFPC from a 1978 environmental study by ORNL. The vegetation samples were confiscated by ORNL on April 12, 1982 and the ORNL employee reprimanded for insubordination. He terminated employment at ORNL in June 1982, believing that his career had been compromised because he had collected unauthorized samples near EFPC (Marshall 1983).

In discussions between the Tennessee Department of Health and Environment (TDHE) and ORO in 1982, the existence of classified reports describing mercury losses from Y-12 was mentioned. These classified reports were then cited by an employee of the State of Tennessee in a newspaper interview. Upon learning of the existence of classified reports on mercury losses from Y-12, as well as the story of the ORNL employee who had conducted unauthorized sampling at Y-12, the *Appalachian Observer* (a local newspaper) filed a Freedom of Information Act (FOIA) request on November 24, 1982 for all reports on mercury spills and emissions at the ORR. In December 1982, the Tennessee State Health Commissioner posted EFPC as unfit for fishing (Marshall 1983) and on May 17, 1983, in response to the FOIA request, a report describing a March 1966 mercury spill at one of the Colex Production facilities (Building 9201-5) and a declassified version of the 1977 mercury inventory report were released to the public (LaGrone 1983).

The release of the declassified version of the 1977 mercury inventory report (Case 1977) generated much public and media interest. When news appeared that more than 2.4 million pounds of mercury had been "lost" or were unaccounted for at Y-12, the plant was deluged with questions.

The 1983 Mercury Task Force Investigation

Several weeks prior to the May 17, 1983 release of the 1977 mercury inventory report, ORO informed the Y-12 plant manager that the declassified version of the 1977 report would be released. Acting on rumors of a Congressional subcommittee hearing to be held that summer, the plant manager asked Y-12 employees to send any mercury documents in their possession to Plant Records. This began the collection of documents that became the Mercury Task Force Files (Wilcox 1995).

On May 20, 1983, three days after release of the 1977 report, the Y-12 plant manager appointed the Mercury Task Force to collect historical data on mercury accountability, study mercury salvage and recovery, and summarize investigations of mercury impacts on worker health and the environment. William J. Wilcox, Jr., Technical Director of Research and Development and the Technical Service Laboratories at K-25 and Y-12, was asked by the Y-12 plant manager to chair the Mercury Task Force. Wilcox had worked at Y-12 in the 1940s in uranium operations, and transferred to K-25 in 1949, where he remained to become technical director in 1969 (Wilcox 1995). The Mercury Task Force consisted of plant employees who were not involved in Colex operations during 1955-1962, when most mercury exposure to workers and losses to the environment occurred (UCCND 1983a).

The first task for the Mercury Task Force involved updating the 1977 estimates of mercury "accounted for", since additional mercury had been removed from process equipment and flasked since January 1977. The Mercury Task Force then reevaluated loss estimates in the 1977 mercury inventory report (UCCND 1983b).

During the sixth week of the Mercury Task Force's investigation, on July 11, 1983, a Congressional subcommittee hearing was held regarding mercury releases from Y-12 operations. The "Hearing on the Impact of Mercury Releases at the Oak Ridge Complex," chaired by Representatives Albert Gore, Jr. and Marilyn Lloyd, was held at the American Museum of Science and Energy in Oak Ridge. Testimony was given by the following individuals and organizations (Lloyd and Gore 1983):

- U.S. Department of Energy, Oak Ridge Operations
- U.S. Environmental Protection Agency
- Tennessee Department of Health and Environment
- Tennessee Valley Authority
- Environmental Sciences Division, Oak Ridge National Laboratory
- Oak Ridge Research Institute
- Michigan State University Institute of Water Research
- City of Oak Ridge Environmental Quality Advisory Board, a voluntary citizen board advising the City Council
- The *Appalachian Observer* newspaper
- The former ORNL employee who took the unauthorized vegetation samples
- Oak Ridge Area Client Council, represented by a Scarboro resident
- Oak Ridge Chapter, NAACP
- Legal Environmental Assistance Foundation, University of Tennessee College of Law
- Mayor of Oak Ridge
- Roane-Anderson Economic Council
- Oak Ridge Chamber of Commerce
- Committee of Fifty, promoting growth and development in Oak Ridge
- Atomic Trades and Labor Council, the employee union at ORNL and Y-12.

On August 18, 1983, the 417-page classified Mercury Task Force Report, *A Study of Mercury Use at the Y-12 Plant, Accountability, and Impacts on Y-12 Workers and the Environment– 1950-1983* (Y/EX-21) (UCCND 1983a) was completed (UCCND 1983c). This report represents the official statement on mercury releases from Y-12. The Mercury Task Force released an updated, unclassified 35-page executive summary (Y/EX-23) in November 1983 (UCCND 1983b). This report included an update on sampling and analyses of sediments in Watts Bar and Chickamauga Lakes, to verify the timing and quantity of mercury losses to EFPC. Two declassified versions of the Y/EX-21 report, denoted Y/EX-24 and Y/EX-21/del rev (UCCND 1983a), were released in December 1983 and March 1994, respectively, due to changes in classification guidelines.

In addition to uses in lithium separation at Y-12, mercury was also used at X-10 and K-25 in much smaller quantities (ChemRisk 1993a). However, mercury operations and releases at X-10 and K-25 were not included in the 1983 Mercury Task Force investigation. As a result of the Mercury Task Force's 1983 consolidation and preservation of historical records on Y-12 Colex operations, more information is available on Colex operations than on other (pilot-scale) lithium separation processes involving mercury at Y-12 (e.g., Orex and Elex) or other mercury uses at Y-12, X-10, and K-25.

3.4 Monitoring Programs for Mercury in Process Releases

Beginning in the early 1950s, monitoring programs for mercury in air and liquid effluents from lithium separations operations were conducted. While operations involving mercury were underway, air monitoring was conducted primarily to protect worker health. Prior to 1970, water monitoring was conducted primarily for material and financial accountability since most of the world's supply of mercury was at Y-12 during 1955-1963, and Y-12 management did not want to lose large amounts of this critical resource. During this time, losses of enriched lithium to EFPC were actually of somewhat higher concern than losses of mercury because enriched lithium was the desired product. After about 1970, environmental monitoring began to focus on off-site mercury contamination.

Historical monitoring and analytical methods for mercury in building air and liquid effluent are described below. Excerpts from historical documents that describe these methods in greater detail are presented in Appendix B of this report.

3.4.1 Monitoring Procedures and Analytical Instrumentation for Airborne Releases

A routine air sampling program for mercury vapor in building air at Y-12 was initiated in 1949. By 1952, reports from the Y-12 Industrial Hygienist indicated that more than 6,000 air samples were collected annually. The data were reported in Health Physics Department reports as percentages of samples having mercury concentrations greater than 0.1 mg m^{-3}— at that time, the acceptable limit for workplace air concentrations. The mercury program was administered in the Y-12 Plant by the joint efforts of the Industrial Hygiene (IH) Section and the Medical Department. The IH section was responsible for monitoring operating areas for mercury vapors and advising area supervision of the air concentrations in their respective areas. Generally, samples from development and production areas where mercury was

handled on a continuing basis were collected on a scheduled basis at predesignated locations. Sampling results were reported routinely to area supervision on a daily, weekly, and/or monthly basis (UCCND 1983a; Patterson et al. 1957; McRee et al. 1965).

When lithium separations operations involving use of large quantities of mercury were being developed at Y-12 (early 1950s), methods for measuring airborne concentrations of mercury were still being investigated by the IH group. Three commercially available methods were identified and, of the three, only the General Electric Instantaneous Mercury Vapor Detector was found to be reliable. The mercury vapor detector measured absorption of ultra-violet light of 2,537 angstrom wavelength by mercury vapor; the absorption is proportional to the concentration of mercury vapor in the atmosphere (UCCND 1983a; McMurray and Redmond 1958; Perry and Napier 1957).

Because the GE instrument was heavy and difficult to use under Y-12 operating conditions, with the very long cord required, a great deal of effort was put into developing a smaller cordless instrument. Such an instrument using DC current was developed and used in the latter parts of the Colex program, from July 1957 to 1962 (McMurray and Redmond 1958). Overall, the majority of building air samples were taken with the portable GE instrument and were of the spot type (i.e., representing the concentration only at the time the sample was taken, not over a prolonged period). The GE instrument and the cordless DC current instrument were used for mercury sampling until 1976. Since 1976, mercury vapor sampling tubes have been used for air sampling (UCCND 1983a). Photograph 1 depicts the portable mercury vapor detector developed by Y-12.

3.4.2 Monitoring Procedures and Analytical Instrumentation for Waterborne Releases

According to the 1983 Mercury Task Force Report, composite samples of EFPC surface water have been collected for laboratory analysis since the early 1950s (UCCND 1983a; Center 1958). These data were used primarily to monitor process losses. With the exception of a period from January 1974 to June 1977, when samples were analyzed for soluble mercury only, all samples were analyzed for total mercury.

Sample collection methods for mercury in EFPC water were as follows:

- From 1951 to 1955, a Y-12-designed trickle sampler was used to collect weekly composite samples of EFPC water. The sampler was designed to collect a 5-gallon composite sample each week. The sample was collected from the top of the stream and did not represent all the suspended particulate matter in the creek. [The 1983 Mercury Task Force applied a correction factor (discussed in Appendix B) to adjust for the effect of collecting samples from the top of the stream (UCCND 1983a)].

- In 1955, a TVA-designed system was installed in EFPC behind the Y-12 warehouse in Building 9720-8. The system consisted of a weir to measure flow

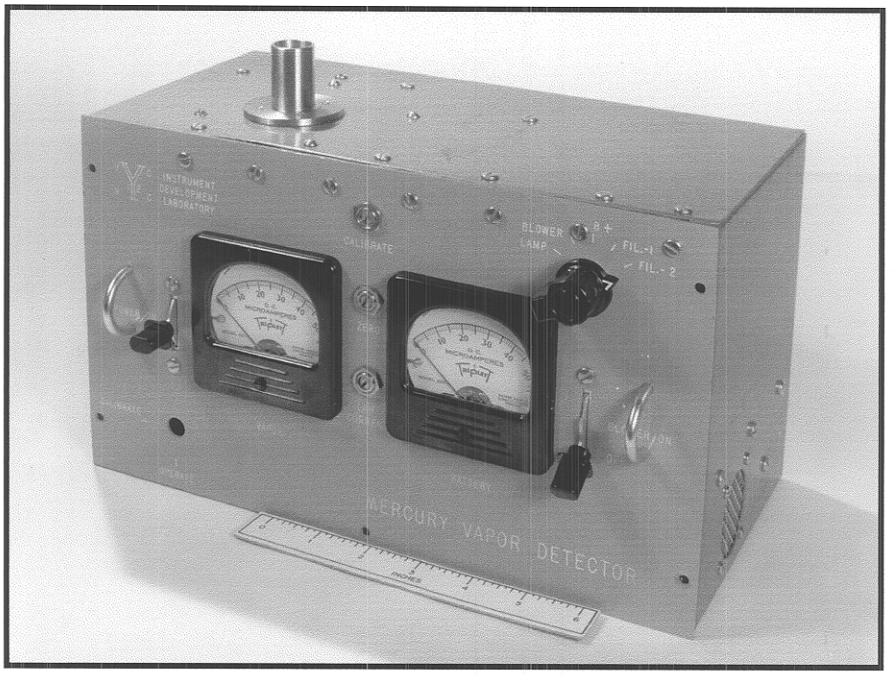
and a tribullar (dipper type) sampler that collected time-proportional, weekly, 5-gallon composite samples (Patterson et al. 1957).

- In 1963, New Hope Pond was constructed, and the sampling point for the weekly composites was moved to the outfall of the pond. A time-proportional sampler was used to fill a 55-gallon drum, from which the weekly composite was taken (McRee et al. 1965).
- Starting in 1973, the weekly composites were poured into a larger bottle to form a monthly composite that was analyzed for mercury and other constituents (UCCND 1983a).
- Starting in 1977, separate grab samples were collected for mercury, and preserved by acidification in the Y-12 Plant laboratory (UCCND 1983a).
- Starting in September 1982, grab samples were acid-preserved in the field at the time of collection (UCCND 1983a).

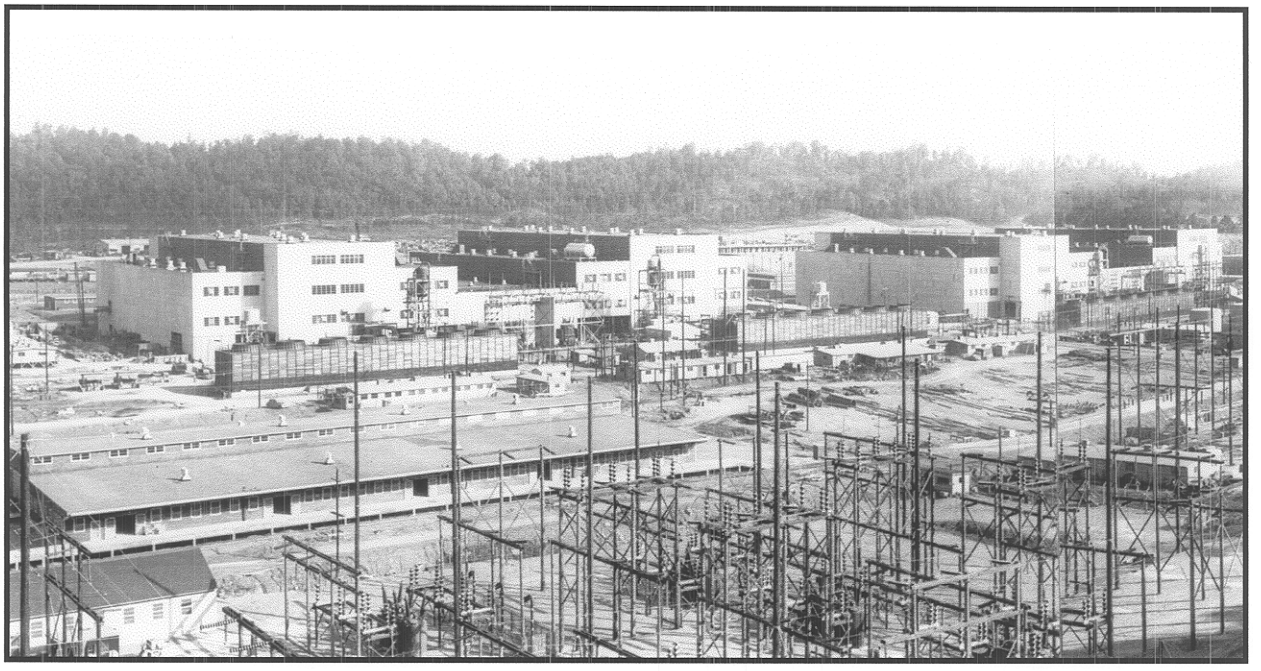
USEPA Method 245.1 for mercury (issued in 1974) recommends preserving water samples by acidification with nitric acid to a pH of 2 or lower at the time of collection, to avoid losses of mercury upon storage. However, since the EFPC composite samples were used to monitor water quality parameters such as pH and bacteria count, acidification would have invalidated the results of the analyses for other parameters. Consequently, composite samples collected in EFPC were not preserved by acidification. Beginning in 1977, however, grab samples for mercury analysis were collected**S** these samples were acidified. Grab samples collected between 1977 and September 1982 were acidified in the laboratory, and samples collected since September 1982 were acidified in the field (UCCND 1983a).

Analytical methods for mercury in surface water were as follows:

- From 1951 until June 1957, the mercury content of EFPC water was determined by a colorimetric technique. A mercury-dithiazone complex was measured spectrophotometrically at 485 nm. This method provided a detection limit of 0.1 mg L^{-1} with a relative limit of error for a single analysis of ± 50% (UCCND 1983a; Fee and Sanders 1982).
- In July 1957, the colorimetric method was replaced by the mercurometer method. All mercury was trapped and converted to the highly insoluble mercuric sulfide, vaporized in a heated chamber, and the mercury detected with a General Electric mercury vapor detector. This method provided a much shorter analysis time, a detection limit of 0.01 mg L^{-1}, and a relative limit of error for a single analysis of ± 40% (McBryde and Williams 1957).
- In August 1967, an atomic absorption method providing a detection limit of 0.001 mg L^{-1} with a relative limit of error for a single analysis of ± 20% was adopted (Dill 1967).



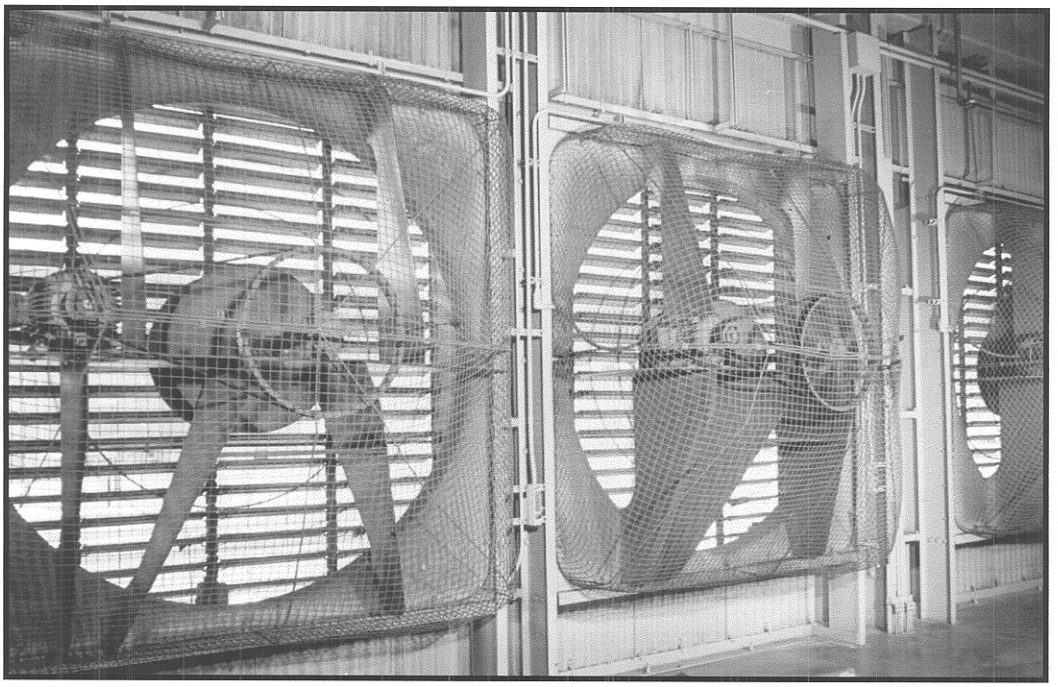
Photograph 1: The portable mercury vapor detector developed at Y-12 in 1957. (Photo 100736 courtesy of the Y-12 Plant)



Photograph 2: Building 9201-5 and 9201-4, originally built to house electromagnetic enrichment operations for uranium during World War II, were later converted to house the Colex process (1945). (Photo 6174 courtesy of the Y-12 Plant)

Photograph 3: Mercury flasks arriving at the Y-12 unloading facility (1955)

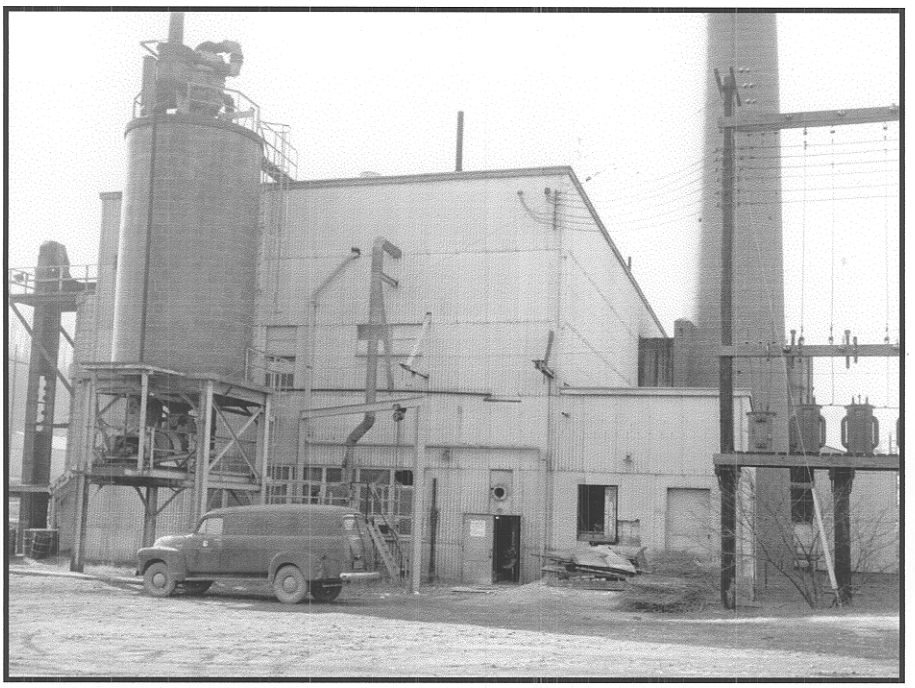
Photograph 4: Workers emptying flasks of mercury at the Y-12 mercury dumping shed. Pipelines carried mercury to the buildings that housed the Colex operations.
(1955 photo ORO-55-762-3 courtesy of U.S. Department of Energy)



Photograph 5: Six-foot diameter fans were installed in Colex Building 9201-5 in 1956 to increase ventilation and thereby reduce airborne mercury concentrations in operating areas (Photo 102455 courtesy of the Y-12 Plant).

Photograph 6: The project team and ORHASP members on a tour of the Alpha-4 Colex production building (1996). Amalgam maker trays can be seen at the lower right of the photo. (Photo 314645 courtesy of the Y-12 Plant)

Photograph 7: Aerial view of Building 81-10, looking northwest (1983). The metal frame structure east of the building is the mercury roaster, which was dismantled after this photo was taken on November 29, 1983.
(Photo 218284 courtesy of the Y-12 Plant)



Photograph 8: Building 9401-2, one of two original Y-12 steam plants (1956). This steam plant was replaced in 1956 by a new steam plant, Building 9401-3.
(Photo 100470 courtesy of the Y-12 Plant).

4.0 SOURCE TERM ASSESSMENT

Source term assessment is the process of reconstructing historical releases of a material from industrial operations to the environment. In reconstructing historical doses or risks to nearby populations, assessors often base estimates of past exposure point concentrations on information describing the mass of a material released from a facility to air or water. Source term assessment supplements historical environmental sampling data, which may not have been collected in all media of interest (such as air, soil, water, vegetation, fish), at the locations where exposures may have occurred, or during the periods of maximum releases.

This section describes the Task 2 source term assessment for mercury releases from the ORR. In particular, this section describes:

- **The document search** and other methods used to locate data on mercury releases from the ORR;
- **Historical data on mercury concentrations in building air and EFPC water, building ventilation rates, and water flow rates**, used to quantify releases from each of the key lithium isotope separation and auxiliary operations at the ORR; and
- **The project team's quantitative estimates of annual mercury releases** to air and water.

Quantitative estimates of mercury releases from the ORR were used to model exposure point concentrations at off-site locations where individuals may have been exposed. Supporting information gathered in the document search and review is provided in Appendices C through F. Tables summarizing the calculations supporting the release estimates are provided in Appendices G through I.

Airborne source terms developed by the project team were based on releases of mercury-contaminated air from the Elex and Colex pilot plants, the Colex production buildings, the mercury recovery facility, outdoor smelting of mercury-contaminated scrap metal, and coal burning at Y-12 and K-25 steam plants. The waterborne source term was based on discharges of mercury-contaminated liquid effluent from lithium separation operations. The lithium separation buildings discharged to a nearby ditch that joined EFPC, and monitoring was conducted in EFPC just past the point of entry of the building discharge lines to the ditch.

A source term for mercury spills to soil was not developed, because approximately half of the spilled mercury was recovered at the time of each spill, and additional mercury was also recovered at the on-site mercury recovery facility from dirt excavated at the spill sites (3.6 million pounds of mercury were recovered at the mercury recovery facility during its operation.) Any mercury runoff to EFPC within the plant boundary and before the sampling location would have been included in the mercury concentrations measured at the site boundary.

Source terms were not developed for other mercury uses at Y-12, K-25, or X-10 due to the small quantities used, or the lack of significant building ventilation, or because information identified by the project team indicated that significant releases (relative to releases associated with Y-12 lithium separation operations) did not occur. Uses for which source terms were not developed at Y-12 included the Orex lithium separation pilot plant, mercury bottling and cleanup campaigns, mercury-thallium alloy use in weapons production, and Y-12 mercury cleaning operations for calutron instrumentation. In addition, source terms were not developed for mercury cleaning operations for gaseous diffusion intrumentation at K-12 or for the Orex, Hermex, or Metallex and other fuel reprocessing operations at X-10. Information collected by the project team on these uses is summarized in Appendix A.

The conclusions of the Task 2 mercury source term investigation for the ORR are summarized below.

Conclusions of the Task 2 Mercury Source Term Investigation

- Airborne releases of mercury were primarily a result of building ventilation installed for worker protection.
- Waterborne releases of mercury were primarily a result of process leaks and spills within buildings that eventually found their way to the storm sewer system, and an early process in which mercury was washed with nitric acid thereby increasing its initial solubility.
- Airborne and waterborne mercury release estimates made by the project team were 43% and 17% higher, respectively, than the estimates made by the 1983 Mercury Task Force.
  - The higher air source term is primarily due to incorporation of information indicating greater ventilation from Building 9201-4 Colex production operations than previously assumed and the inclusion of releases from Building 9204-4 Elex production operations.
  - The higher water source term is primarily due to reestimation of releases in 1953-1955.
- Pilot plant operations and other minor uses of mercury at Y-12, K-25, and X-10 were found not to be sources of significant releases of mercury relative to mercury releases from Y-12 Colex production facilities. The lower magnitude of these releases estimates was due to use of smaller quantities of mercury and/or minimal ventilation, or no evidence of release.

4.1 Identification and Review of Historical Data on Mercury Releases from the ORR

A primary focus of Task 2 of the Oak Ridge Dose Reconstruction was investigating historical data on mercury releases to air, water, and the ground from the ORR and comparing these data to previously reported estimates of mercury releases. Previous estimates include estimates by Y-12 personnel (Case 1977) and the 1983 Mercury Task Force (UCCND 1983a,c).

The investigation of mercury releases from the ORR included:

- Review of the 1977 Y-12 Mercury Inventory Report (Case 1977);
- Interviews with members of the 1983 Mercury Task Force;
- Review of the classified and unclassified versions of the 1983 Mercury Task Force Report (UCCND 1983a,c);
- Review of retired files in the Y-12 Records Center and the 1983 Mercury Task Force Files; and
- Review of retired files at K-25 and X-10.

In addition, a summary of data needs for the Task 2 investigation was distributed to all Dose Reconstruction project task managers and Task 5 personnel (Task 5 was the Dose Reconstruction document search task, which focused on systematic review of record holdings at all Y-12, K-25, and X-10 document repositories). Most of the historical literature uses code words to describe the materials used in lithium separation processes. "Alloy" was the term used for lithium, mercury was called "solvent", and the isotopes of ^{6}Li and ^{7}Li were referred to as "aspen" and "marble", respectively (Code Words 1962). Document searches conducted by the Task 2 team for information on these materials included these former code words as keywords.

4.1.1 Review of the 1977 Mercury Inventory Report

The 1977 Mercury Inventory Report (Case 1977) was the first effort by Union Carbide to reconstruct the historical mercury inventory at Y-12. The 1977 Mercury Inventory Report stated that "2.4 million pounds of mercury has either been lost [to the environment] or unaccounted for [a difference between the quantity originally received and the mercury that can be quantitatively described] ." The distinction between "lost" and "unaccounted for" arose because mercury lost to air and water and released to off-site locations can be estimated by effluent monitoring, and this quantity can therefore be "accounted for" as lost to the environment. On this basis, the 1977 Mercury Inventory Report estimated 1.9 million pounds of mercury were unaccounted for, by subtracting weighed quantities (accounted for) and measured releases (lost) from the total quantity of mercury vouchered to Y-12. Air, water, and spill losses to soil were estimated at 557,000 pounds (UCCND 1983b).

Comparisons of the mercury material balances provided in the 1977 Mercury Inventory Report to estimates provided in the later 1983 Mercury Task Force Report are presented in Section 4.1.3.

4.1.2 Interviews with Members of the 1983 Mercury Task Force

Four key members of the 1983 Mercury Task Force were interviewed by the Task 2 team concurrent with review of the August 18, 1983 Mercury Task Force Report and the Mercury Task Force Files:

- Mr. William J. Wilcox, Jr., Mercury Task Force Chairman
- Mr. John M. Napier, Mercury Task Force Consultant and 1977 report co-author
- Mr. David W. Smith, Mercury Task Force Consultant and 1977 report co-author
- Mr. Lowell L. McCauley, Mercury Task Force Worker Health Chairman

Samples of the questions used to interview Mercury Task Force members are listed in Appendix C. Copies of the interview notes are retained in the project repository (Repository Numbers 1668, 1673, 2008, 1671, and 3270).

4.1.3 Review of the 1983 Mercury Task Force Report

The Task 2 team reviewed both the unclassified and classified versions of the 1983 Mercury Task Force Report (UCCND 1983a,c), as well as the updated executive summary of the 1983 Mercury Task Force Report (UCCND 1983b). The 1983 Mercury Task Force Report is the most detailed discussion of mercury releases from Y-12, and represents the official record of releases of mercury from the Y-12 Plant. Only the unclassified version of the 1983 Mercury Task Force Report is discussed here, to allow general publication of the current report. The classified version contains details about process operations not directly related to mercury releases from Y-12.

The six sections of the 1983 Mercury Task Force Report are listed below with brief descriptions of section contents.

Outline of the 1983 Mercury Task Force Report

Section 1– History of Lithium Isotope Separation at Y-12, describing technical operations.

Section 2– The Receipt of Mercury at Y-12, describing receiving operations and estimates of the total quantity of mercury originally vouchered to Y-12.

Section 3– Quantities of Mercury Accounted For by Y-12, describing bottling operations, quantities of mercury in process waste, remaining mercury inventory in buildings and storage, and quantities of mercury recovered and sold.

Section 4– Quantities of Mercury Lost or Not Accounted For by Y-12, describing mercury lost to air, water, and spills, mercury under buildings and in drain systems, and theft.

Section 5– Studies of the Health of Y-12 Employees Exposed to Mercury and Review of Health Protection Programs, describing historical mercury exposure guidelines, the urine monitoring program for workers, the 1974 study of workers who were exposed in the 1950s, the 1983 worker mortality study, and programs to reduce worker exposure.

Section 6– The Environmental Impacts of Mercury Releases and Losses, describing environmental sampling, including a summary of all studies conducted prior to 1983.

The remainder of this discussion covers information from Sections 2, 3, and 4 of the 1983 Mercury Task Force Report relevant to releases of mercury to the off-site environment. Section 5 of the 1983 Mercury Task Force Report summarizes studies conducted in an effort to understand and control sources of mercury exposure for workers, including building ventilation changes. Changes in building ventilation are relevant to mercury release estimates because increasing ventilation to reduce indoor exposures probably increased releases of airborne mercury to outside air. Section 6 of the 1983 Mercury Task Force Report summarizes studies that measured mercury concentrations off-site**S** these are included in the discussion of off-site environmental monitoring in Section 5 and Appendix J of the current report.

Comparison of 1977 and 1983 Estimates of Mercury Material Balances

The project team requested declassification of all pages discussing total mercury quantities at Y-12 from the original classified versions of the 1977 Mercury Inventory Report and the 1983 Mercury Task Force Report, to update publicly available information. This request was satisfied by the release of the Y/EX-21/del rev report (UCCND 1983a). The project team reviewed the basis for the total mercury quantities presented in these reports, and verified that the figures from inventory vouchers and reported weights were accurately recorded.

The total quantity of mercury vouchered to Y-12 was recently declassified (USDOE 1993). The 1983 estimate of the quantity vouchered to Y-12 was 28,000 pounds greater than the 1977 estimate; however, the total remained approximately the same at 24.3 million pounds (USDOE 1993).

As shown above, the 1977 Mercury Inventory Report reported that 1.9 million pounds of mercury were unaccounted for and 557,000 pounds were lost to air, water, and spills, for a total of about 2.4 million pounds of mercury lost or unaccounted for (UCCND 1983b). Revised estimates in the 1983 Mercury Task Force Report were 1.3 million pounds of mercury unaccounted for, 733,000 pounds lost to air, water, and spills, yielding a total of about 2 million pounds lost or unaccounted for.

Table 4-1 compares the 1977 and 1983 estimates of mercury material balance. Differences between the 1977 and 1983 estimates of mercury material balances at Y-12 are as follows:

- The estimated total lost or unaccounted for decreased from 2.4 million pounds in the 1977 report to 2.0 million pounds in the 1983 report, primarily due to increases in "accounted for" categories. For example, mercury removed from process equipment and bottled between 1977 and the 1983 investigation increased the "accounted for" estimate (UCCND 1983b).
- The estimated losses to air increased from 30,000 pounds in the 1977 report to 51,300 pounds in the 1983 report (UCCND 1983b).
- The estimated losses to EFPC decreased from 470,000 pounds in the 1977 report to 239,000 pounds in the 1983 report because an error in the 1977 estimate was found during the 1983 investigation. The 1977 estimate was high by a factor of two because, for a brief time around 1977, water samples were filtered and only insoluble mercury was measured. Investigators in 1977 erroneously assumed that water samples had always been filtered and doubled their final estimate of quantities of mercury lost to EFPC to account for soluble and insoluble mercury (UCCND 1983b).
- The estimated losses to spills increased from 50,000 pounds in the 1977 report to 425,000 pounds in the 1983 report, due to the addition of seven spills (UCCND 1983b).

The 1983 Mercury Task Force speculated on the location of about half, or 645,000 pounds, of the "unaccounted for" mercury. Ten percent was estimated to have been in production buildings and the remainder was believed to reflect over-estimation of quantities received at Y-12, due to inadequate record keeping. The only records of the quantity of mercury received at Y-12 were transfer vouchers. The units of measure on these vouchers were numbers of 76-pound flasks, not weight in pounds. Because of the urgency of lithium enrichment operations in the 1950s, mercury flasks received at Y-12 were not weighed, and witnesses to the 1950s mercury receiving operations claim that some flasks were only partially full and may have leaked prior to arrival at Y-12. In 1957, when unopened mercury flasks were shipped off-site, Y-12 received complaints of leaking flasks and shortages from the off-site recipients. The 1983 Task

Force estimated that the quantity of mercury vouchered to Y-12 was overestimated by about 0.5 million pounds. There are no data to support the estimates of mercury remaining in building structures and mercury never received, so the Task Force did not include these estimates in "accounted for" quantities (UCCND 1983b).

Table 4-1: 1977 and 1983 Mercury Material Balance Estimates by Y-12 Plant Staff

Source of Material Inventory and Losses	1977 Mercury Inventory Report (Case 1977)	1983 Mercury Task Force Report
VOUCHERED to Y-12:	24,321,000	24,348,852
Returned unopened or rebottled and stored/sold	*	21,666,348
In lithium hydroxide tails, sold and stored	1,000	1,400
In Building 9201-5 scrap, sold	10,000	14,000
In Building 9201-5 sludge, removed and sold	111,000	174,000
As flasking overage given to GSA	12,000	17,212
In Building 9201-4 equipment, still in place	*	200,000
In sludges and sumps in Alpha-4 Building	100,000	250,000
In Building 9201-2 sewer pipe	**	800
ACCOUNTED FOR Total:	*	22,323,796
Known LOST and NOT ACCOUNTED FOR Total:	2,437,752	2,025,056
Known lost to air	30,000	51,300
Known lost to East Fork Poplar Creek	470,000	238,944
Known lost to New Hope Pond sediment, Chestnut Ridge	7,200	6,629
Known lost to New Hope Pond sediments now in place	**	8,475
Known lost to ground, Building 9201-5 spill accident	49,853	49,853
Known lost to ground, seven other spills	**	375,000
Known lost to ground, Building 81-10 operations	**	3,000
Known LOST Total:	557,053	733,201
NOT ACCOUNTED FOR Total:	1,880,699	1,291,855

* These data were classified for security reasons in 1977.

** Data not available in 1977 report.

The numbers from the report are probably not accurate down to the one pound level. However, the exact values were retained for accounting purposes, according to the 1983 Mercury Task Force Report.
Source: UCCND (1983a).

The project team did not use the estimates of quantities lost that are described above in their assessment of off-site releases and doses. Estimated mercury losses were recalculated by the project team using documents located during the document search and references from the 1983 Mercury Task Force Report. Documents collected by the project team included reports of building air concentrations and ventilation rates in Colex buildings and surface water concentrations and flow rates in EFPC.

4.1.4 Review of Y-12 Mercury Task Force Files and Retired Files

The Y-12 Mercury Task Force Files contain the documents regarding mercury use at Y-12 assembled around June 1983 during the Mercury Task Force investigation (UCCND 1983a). Boxes of retired files in the Y-12 Records Center, not physically moved into the Mercury Task Force Files in 1983, also contain information on historical mercury operations at Y-12. The project team's review of the Mercury Task Force Files and the Y-12 Records Center retired files is described below.

Mercury Task Force Files

The Y-12 Mercury Task Force Files were assembled during the Mercury Task Force investigation in response to a plant-wide request from the Y-12 plant manager (UCCND 1983a). The Task 2 team located a copy of this request, the *Records Management Directive**S** Mercury* (May 16, 1983), in the Mercury Task Force Files. The letter states:

> "The Y-12 Plant Records Department has been directed to place an immediate freeze on the destruction and/or transfer of all records related to the subject of mercury... In the following weeks, the Plant Records Department will be establishing an information system involving all records related to this subject. In order for this to be complete, it is necessary that each of you [to] notify appropriate personnel in your division to report all records on hand regarding the subject of mercury to the Y-12 Records Officer, W.D. Minter ... no later than June 1, 1983..."

The Mercury Task Force Files are stored in four classified safes in a vault in Building 9711-5. All documents in the files were originally classified as Secret or Confidential Restricted Data (SRD or CRD) so that the large number of historical documents that were unmarked (with regards to classification) would not have to be reviewed by the Y-12 Classification Office and because classified information was contained in many technical documents and Y-12 Plant Quarterly Reports. Mercury Task Force did not have the time nor the need during the 8-week investigation to separate unclassified documents from classified documents, or to produce sanitized versions of classified documents, particularly since original documents were to be preserved intact (Wilcox 1995). As such, Mercury Task Force headquarters were inside the security fence, and all Mercury Task Force members had Q-clearances. Currently, a Q-clearance and "need to know" are required for access to the files.

The drawers in the four classified safes contain Mercury Task Force File folders numbered M1 through M853. There are no documents in the files dated after June 1983. Multiple folders exist for many M-numbers because, during the 1983 investigation, small documents related to the same topic or submitted

together were given the same M-number. As of 1983, the files contained a total of 1,228 documents. Appendix D of this report lists the M-files located in the Mercury Task Force Files.

During the 1983 investigation, Mercury Task Force File documents were listed in an electronic database; however, the database in electronic form was subsequently lost (HAI 1994). Several alternative approaches can be used to identify Mercury Task Force File documents:

- Two database printouts remain in the Mercury Task Force Files: one numerical by M-number and the other alphabetical by title. An unclassified version of the numerical database printout is now available to the public as a result of the Task 2 team's review of the files.

- Since M-numbers sometimes relate to specific subjects, the M-number of a document released to the public in 1994 as part of the Large-Scale Review of classified documents can be used to locate related documents in the Mercury Task Force Files. For example, M-487 documents contain information on the Solvent (Mercury) Hazard Committee actions to reduce mercury air concentrations in Colex buildings, and M-843 documents contain the Elwood (1977) study and related correspondence.

- The 121-page bibliography of the 1983 Mercury Task Force report (UCCND 1983a) describes the 1,228 documents that were located in the Mercury Task Force Files in 1983. An unclassified version of this bibliography is available to the public. However, the bibliography is organized by document type (i.e., correspondence, progress reports, accounting data, health records, and open literature) rather than by M-number. When the Mercury Task Force Files were inventoried as classified documents in 1985, they were rearranged by M-number instead of document type to facilitate efficient location of documents.

- Appendices D and E of this report contain spreadsheets listing the contents of the Mercury Task Force Files, sorted by M-number, and the Mercury Task Force Files available to the public in the DOE-Oak Ridge Public Document Reading Room.

Activity relating to the Mercury Task Force Files since 1983 has included requests for copies of documents relevant to pending lawsuits, Freedom of Information Act (FOIA) requests, the Large-Scale Review declassification effort, and identification of epidemiologic records (McCauley 1995).

Over 414 documents (as of August 14, 1995) from the Mercury Task Force Files are available to the general public in the DOE-Oak Ridge Public Document Reading Room (55 Jefferson Circle, Oak Ridge, TN) as a result of the Large-Scale Review program at Y-12, completed in September 1994 (Fraser 1995). The Large-Scale Review of classified documents at Y-12 supported USDOE efforts to provide

environmental, safety, and health information to the general public, comply with the USDOE/Tennessee Oversight Agreement, and declassify or downgrade document holdings (Keyes 1994). The Large-Scale Review began with review of the Mercury Task Force Files by the Y-12 Classification Office, due to the anticipated needs of the Dose Reconstruction project for mercury-related documents.

The Large-Scale Review project set out to declassify more Mercury Task Force Files documents than were eventually released to the public. Each of the documents that were thought to be unclassified during the initial review was assigned a "Y/HG-" number. However, the publicly-available collection includes only documents that could be declassified in their original form; no pages were deleted or text removed to facilitate declassification. As such, while Y/HG- numbers 1 through 549 were assigned to documents, only 414 Y/HG documents were found to be unclassified after a second review and were released to the public as of August 14, 1995. Appendix E of this report lists publicly-available Y/HG- documents.

A moratorium on destruction of records relevant to epidemiologic and health-related studies has existed since 1989. In June 1994, History Associates Inc. (HAI) reviewed the Mercury Task Force Files as part of USDOE's Epidemiologic Records Inventory Project to verify and inventory epidemiologic and health records at USDOE and USDOE contractor sites. HAI conducted the pilot study at the ORR to assist USDOE in providing information requested by the State of Tennessee and other interested parties in a March 1994 meeting. The Task 2 team examined the HAI report (HAI 1994) identifying documents relevant to epidemiologic and other health-related studies during their initial review of the Mercury Task Force Files.

The Mercury Task Force Files record descriptions given to the record series, as shown in Appendix D, are as follows:

- *Mercury Inventory*– original shipments, inventory calculation worksheets, and accounting documents
- *Mercury Flasking*– correspondence regarding building draining and stripping operations to recover mercury and rebottle it for storage
- *Mercury Storage*– storage buildings, inspections, and other issues related to the storage of mercury at Y-12
- *Mercury Shipments*– shipping orders and transfer forms, bids, public sale
- *Mercury Environmental*– environmental monitoring data (air, water, sediments, fish), reports, and related correspondence
- *Y-12 Production/Operations*– building, process and equipment information that does not contain information on quantities of mercury used or released from Y-12

- *Y-12 Plant Quarterly Reports (4Q52 to 4Q81)*– early years of the series contain quarterly averages for mercury concentrations in building air and quarterly averages for concentrations of mercury in EFPC

- *Technical Division Monthly Progress Reports (1-55 to 12-58)*– contain monthly averages of mercury concentrations in EFPC

- *Quarterly Technical Progress Reports (3Q59-4Q59, 2Q63-3Q63, 3Q64, 2Q66-3Q66, 1Q67, 3Q67-2Q68, 2Q71, 2Q75)* – continuation of the series above, but do not contain EFPC mercury concentrations

- *Alloy Division Weekly Progress Reports (1-55 to 3-55, 5-55 to 3-59)*– weekly highlights of lithium separation operations

- *Alloy Division Monthly Progress Reports (10-55 to 12-55, 2-56 to 12-61)*– monthly compilation of the reports described in the series above

- *Technical Reports (1947-81)*– bound reports on technical topics given a formal report number and distributed outside of Y-12

- *Technical Memoranda (1953-83)*– unbound reports on technical developments, including experiment results and Colex pilot plant runs

- *Health Physics-Hygiene Progress Reports (1-49, 5-49, 4Q50, 6-51 to 12-51, 1-52 to 12-53)*– contain some mercury concentrations in building air

During the Task 2 review, Q-cleared members of the project team had unlimited and unescorted access to the Mercury Task Force Files. To document information gathered on each M-file number during their review of the Mercury Task Force Files, the project team used the Appendix D spreadsheet. The spreadsheet contains a brief description of the contents and date of each file, followed by columns indicating whether the file was listed in the June 1983 Mercury Task Force database printout, whether the file was identified by HAI as relevant to dose reconstruction, and Y/HG- or Y/EXT- document numbers created from the file. Y/EXT- documents are extracts of classified documents that the project team requested during the review. The last column in the spreadsheet indicates whether material from the file was copied for potential use in reconstructing source terms, and subsequently entered into the Task 2 team's database.

The Task 2 project team identified 109 of the 853 M-files as missing or as empty folders. Of these, 39 M-file numbers were missing from both the Mercury Task Force Files and the June 1983 database printout, and may have never been assigned or been initially reserved for documents and then not used. The remaining 70 missing files are described on the June 1983 database printout. The project team submitted a list of missing file numbers and their contents, as described in the database printout, to Y-12 personnel along with a request for any information on the files' fate or location. Based on interviews

conducted by the project team, it appears that some people may have refused to submit their copies of the documents to the files such that the documents may have never actually been in the files.

The status of the 70 missing files is as follows:

- 1 file is listed as missing in the June 1983 printout;
- 21 files are mercury air and water data contained in Health Physics boxes previously reviewed by Task 5 team members at the Y-12 Records Center;
- 7 files are duplicates of other files located in the review of the Mercury Task Force Files (these may have been database entry errors);
- 20 files are available in the open literature in journals, textbooks, or published ORNL or USEPA reports; and,
- 21 files are unexplained (i.e., listed in the printout but not in the files).

In addition to the 414 Y/HG-documents made available to the public by the Large-Scale Review project, the Task 2 project team's review of the Mercury Task Force Files resulted in the release of additional documents to the public reading room:

- Y/EX-21/del rev**S** An unclassified version of the 1983 Mercury Task Force Report based on the most recent USDOE classification guidelines for lithium enrichment operations and the USDOE openness initiatives regarding quantities of mercury used at the Oak Ridge Reservation. This version contains more information than the previous unclassified version, Y/EX-24.
- The June 1983 Mercury Task Force Files database printout (in numerical order)
- Extracts of Y-12 Quarterly Reports related to environmental mercury concentrations
- Extracts of Technical Division Progress Reports related to environmental mercury concentrations
- Sanitized versions of some M-file documents and Y/HG- documents that failed the Large-Scale Review project criteria but were needed by the project team

Y-12 Retired Files

A manual of Radiation Safety records stored at the Y-12 Records Center was prepared for the Radiation Safety Department by Records Management in 1981 (Sykes 1981). This 36-page manual describes the contents of 69 boxes, covering the period 1943 to 1978, submitted to Plant Records by the Y-12 Health Physics Department. The Task 2 team reviewed this manual and identified 33 of the 69 boxes as potentially containing information relevant to estimating mercury releases. These boxes address mercury operations at Y-12 between 1951 and 1963. Some contain raw data from building air monitoring and surface water sampling for mercury. Check-out cards were found in several boxes, identifying files removed by the Mercury Task Force in 1983.

Appendix F lists the boxes containing mercury air and water monitoring data that were located at the Y-12 Records Center.

4.1.5 Review of X-10 and K-25 Retired Files

Documents describing mercury operations at X-10 and K-25 were located during the Task 5 document search at X-10 Laboratory Records in Building 4500N, and at the K-25 Site Records Center in Building K-1034-A. Detailed descriptions of mercury operations at the X-10 and K-25 sites, including estimated inventories and releases of mercury, are presented in Appendix A.

4.1.6 Conclusions from the Task 2 Document Search

Approximately 250 of over 1,200 documents in the Mercury Task Force Files contain information on mercury that is relevant to dose reconstruction. Very few of these documents are classified. The Task 2 team compiled air and water mercury concentration data and other information on building ventilation rates and EFPC flow rates from these records as well as from approximately 50 boxes at the Y-12 Records Center, to reconstruct source terms for the release of mercury from the ORR to the environment. Several documents related to the use of mercury at Y-12 were made available to the public as a result of the document search.

4.2 Information Describing Mercury Releases

A number of information sources that could be used to describe and quantify mercury releases to air and water from Y-12 lithium separation operations and supporting processes were identified by the Task 2 team during review of historical records. The following sections summarize the types of process-related mercury releases to the environment that occurred, and describe the record series that provide information that can be used to quantify the mercury losses.

4.2.1 ORR Operations Associated with Mercury Releases

Operations that resulted in process-related mercury releases to air included:

- **Process leaks and spills within buildings**. At ambient temperatures, elemental mercury is a liquid and has a relatively high vapor pressure, a measure of its potential to volatilize following leaks and spills. In particular, there were many leaks and several large spills of liquid mercury associated with the Colex process due to the large quantities of mercury used in the process and the high pressure of the system (UCCND 1983a).

- **External venting of hydrogen as a by-product of the amalgam-making process**. At the end of the separation step, the amalgam was decomposed to release the ^{6}Li to a water phase. Water and the amalgam were introduced into devices called decomposers, which were filled with crushed graphite. When the water contacted the amalgam, lithium in the amalgam reacted with the water to form lithium hydroxide and hydrogen gas. During Elex operations, the hydrogen gas, contaminated with mercury, was not treated to remove the mercury, and was therefore a source of mercury release to air. However, in the Colex process, the hydrogen gas was scrubbed to remove mercury (UCCND 1983a).

- **Processing of sludge in the Building 81-10 furnace.** "Sludge" was a term used at Y-12 for salvage materials that contained mercury, such as filter solids, cleanings from storage tanks and floor drains, used graphite from the decomposers, and gravel and dirt recovered from spills to the ground. Many of these sludges, including materials from the Colex process, were processed in the Building 81-10 furnace to recover mercury for reuse. Mercury was released to air from operation of the furnace due to incomplete condensation of mercury vapor. Some sludges were sold to scrap dealers, who processed the material off-site to recover metallic mercury (UCCND 1983a).

- **Equipment disassembly**. Disassembly of process equipment after lithium separation operations ceased resulted in the generation of large quantities of scrap metal contaminated with mercury. In one instance, two scrap dealers set up a furnace on site to process metal for sale. This smelting operation resulted in releases of mercury to air (UCCND 1983a).

- **Coal burned in steam plants.** Large amounts of electricity, generated by the Y-12 steam plants, were required at several steps in the isotope separation process. Devices known as amalgam makers, used to produce the lithium-mercury amalgam, required very large amounts of electricity. In Elex, large amounts of electricity were used to keep the lithium in the mercury and the amalgam from

decomposing. Large amounts of electricity were also needed to run the pumps used in the lithium separation process. The coal burned in the steam plants contained small amounts of mercury, which resulted in air releases of mercury.

Process-related mercury releases to water included:

- **Process leaks and spills within the buildings**. Losses to water were primarily through overflow of sumps and spills and leaks that eventually found their way to the storm sewer system (UCCND 1983a).

- **Acid washing of mercury.** Impurities in the water or in the mercury used in the separation process were detrimental to the process, and resulted in reduced production of ^{6}Li. As a result, all water was demineralized, and mercury was washed with a nitric acid solution to prevent the accumulation of impurities during the separation process. Washing of the mercury with nitric acid resulted in the production of mercuric nitrate, a very soluble form of mercury. The nitric acid wash was believed to be the cause of solubilized mercury being released to EFPC, and was therefore replaced by another process in June 1958. A housekeeping program designed to minimize worker exposure to airborne mercury through prevention and clean up of spills also likely resulted in increased generation of mercury-contaminated water. Sodium hypochlorite (bleach) was used to mop floors and clean up spills, since it reduced the volatilization of mercury. Unfortunately, it also increased the solubility of mercury and increased mercury releases through floor drains to EFPC (UCCND 1983a).

Figure 4-1 is a drawing of lithium facilities at Y-12 as they existed in the 1950s and early 1960s. Figure 4-2 is a time line of Y-12 historical operations that were the sources of mercury releases.

4.2.2 Sources of Building Air Monitoring and Ventilation Rate Data

Very little information is available on stack releases of mercury from Y-12. Most of the mercury releases to air were believed to have been due to ventilation of lithium isotope separations buildings to reduce worker exposures. Thus, to estimate air emissions from Y-12, data on Y-12 building air concentrations and building air flow rates during the peak production years, including information on ventilation equipment parameters (such as diameters of fans and stacks, fan speeds, and air volume flow rates), were required. The Task 2 team located data describing Y-12 building air concentrations and building air flow rates in publicly available documents in the DOE Reading Room, in the 1983 Mercury Task Force Files, in boxes archived at the Y-12 Records Center, or in records provided by members of the Y-12 Engineering Drawings Group. Sources of historical data describing mercury concentrations in building air and building ventilation parameters are listed in Table 4-2.

Source: Y/EX-21/del rev (UCCND 1983a)

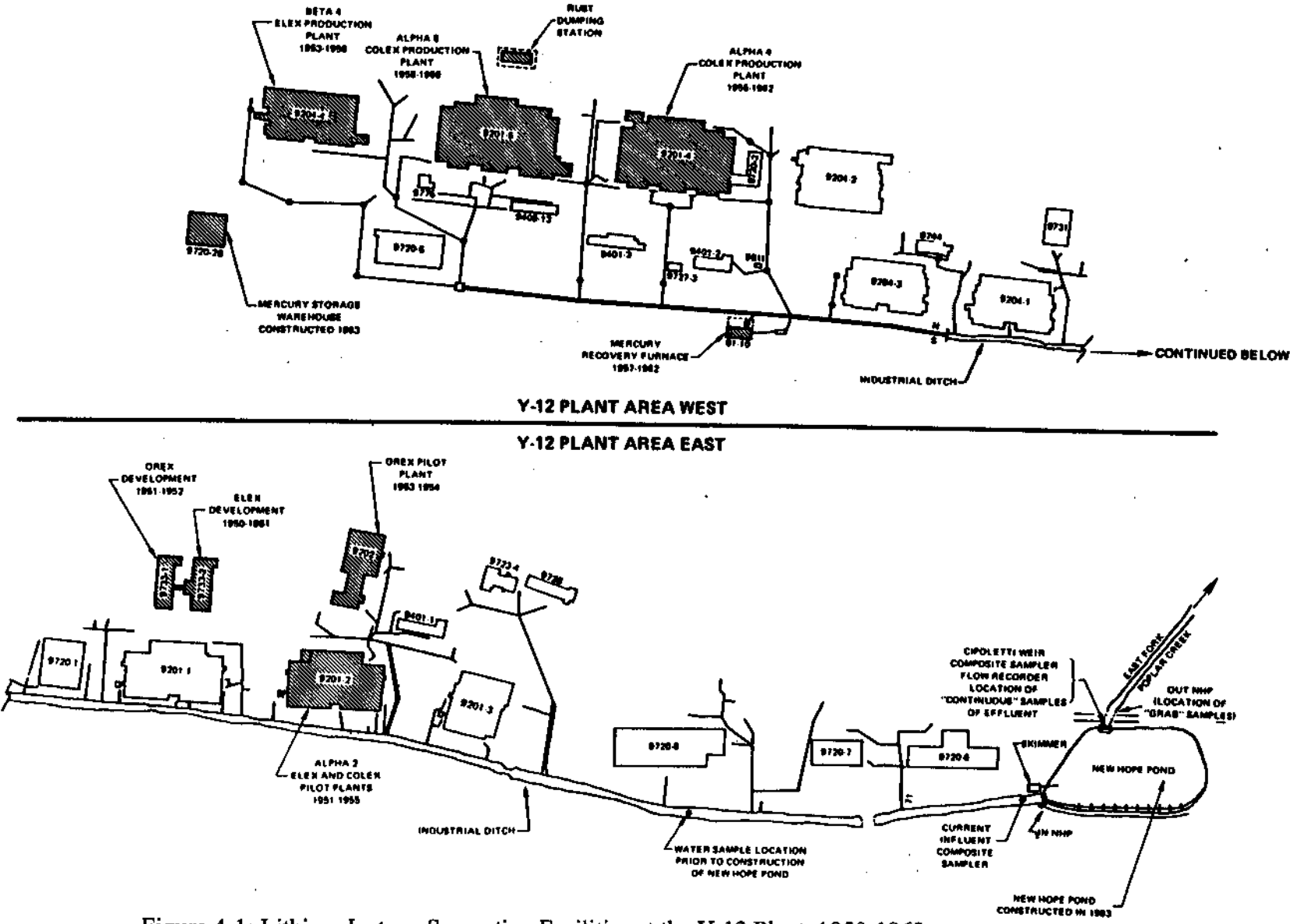


Figure 4-1: Lithium Isotope Separation Facilities at the Y-12 Plant, 1950-1963

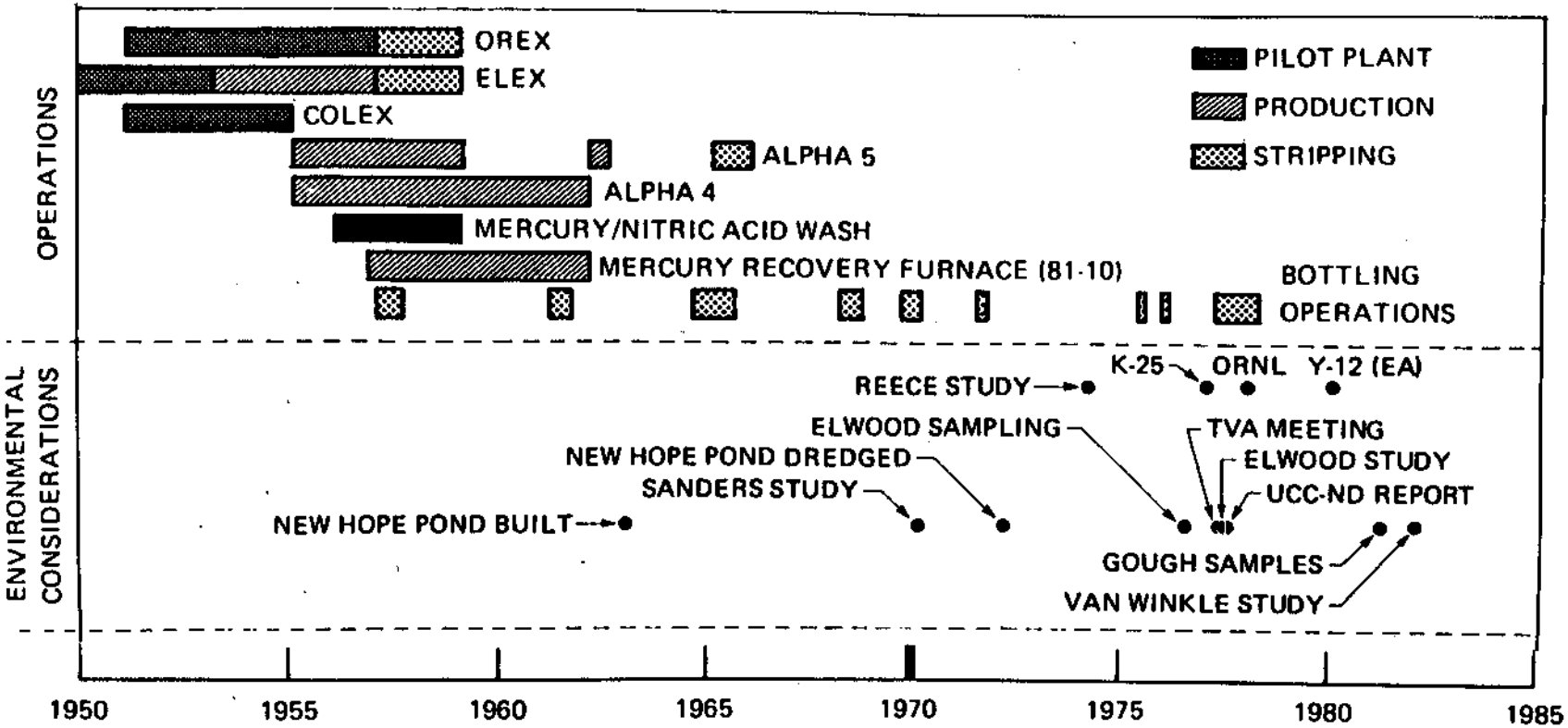


Figure 4-2: Time line of lithium isotope separation operations at the Y-12 Plant and related events. [Source: Y/EX-21/del rev (UCCND 1983a)]

Table 4-2: Sources of Historical Data on Mercury Concentrations in Building Air and Building Ventilation Rates

Building Air Concentration Data

- **Weekly Industrial Hygiene Mercury Air Analysis Reports** (ChemRisk Repository No. 3262). These reports present weekly-average building air concentrations in lithium separation pilot plants for 1953-54.
- **Health Physics Solvent Reports** (ChemRisk Repository No. 3262). These reports present quarterly, monthly, and weekly-average air concentrations in Building 9201-4 for 1954-55.
- **Preliminary Report on Personnel Exposure to Mercury in the Colex Plants** (ChemRisk Repository No. 3262). This report presents monthly-average air concentrations in Colex production buildings 9201-4 and 9201-5 for 1955-57 (LaFrance 1957).
- **Monthly Industrial Hygiene Solvent Air Analysis Reports** (ChemRisk Repository No. 3262). These reports present monthly-average air concentrations in all lithium separation buildings for 1955-60.
- **Y-12 Plant Quarterly Reports** (ChemRisk Repository No. 3264). These reports present monthly-average building air concentrations in Colex production buildings 9201-4 and 9201-5 for 1955-61.
- **1983 Mercury Task Force Report** (ChemRisk Repository No. 3272). This report presents quarterly-average building air concentrations and quarterly estimates of pounds of mercury released to air from Colex production buildings 9201-4 and 9201-5 for 1955-62 (UCCND 1983a).

Building Ventilation Data

- **Baumann 1952** (ChemRisk Repository No. 3262). This report presents a ventilation study for Building 9201-2.
- **Baumann 1953a** (ChemRisk Repository No. 3263). This report presents a ventilation study for Building 9204-4.
- **Little 1956** (ChemRisk Repository No. 3263). This report presents a ventilation study for Building 9201-5.
- **Y-12 building drawings** from the Y-12 Engineering Drawings Group (Choat 1996) (ChemRisk Repository No. 3263)

Y-12 building ventilation system drawings by E.E. Choat, a former Y-12 HVAC (heating, ventilation, and air conditioning) engineer and consultant to the project team, were used to estimate air flow rates from the lithium isotope separations buildings (Choat 1996; Appendix G). Choat estimated building air exhaust flow rates for Buildings 9201-5, 9201-4, 9204-4, 9201-2, 9401-1,-2 and -3, and 81-10 using Y-12 engineering drawings. Choat also provided estimates of release point height, cross-sectional area, exit velocity, and temperature required for air dispersion modeling of mercury releases for these facilities. Information on air releases of mercury from burning coal at the Y-12 steam plants was obtained from a 1991 ambient air monitoring report (Turner et al. 1991).

4.2.3 Sources of Water Monitoring and Flow Rate Data

The Task 2 team located historical data from the Y-12 liquid effluent monitoring program in publicly available documents in the DOE Reading Room, in the 1983 Mercury Task Force Files, in boxes archived at the Y-12 Records Center, or in records provided by members of the Y-12 Health, Safety, Environment and Accountability (HSEA) Division Surface Water Group. In addition to water concentration data, water flow rates measurements were required by the project team to estimate the pounds of mercury released to EFPC. Sources of historical data describing mercury concentrations in liquid effluent released to EFPC and water flow rates are listed in Table 4-3. Each of these sources was reviewed to develop the source terms for mercury releases to EFPC used in the dose reconstruction.

4.3 Quantification of Releases from Y-12 Processes and Facilities

This section describes the Y-12 processes and facilities for which the project team calculated source terms for mercury releases to the environment. These processes and facilities are:

- the Colex Pilot Plant (Building 9201-2),
- the Elex Production Scale Facility (Building 9204-4),
- the Colex Production Scale Facilities (Buildings 9201-5 and 9201-4),
- the Mercury Recovery Facility (Building 81-10), and
- the Y-12 Steam Plants (Buildings 9401-1, 9401-2, 9401-3).

Section 4.4 discusses the data and methodology used by the project team to estimate airborne releases of mercury from these processes and facilities, and compares these figures to estimates developed by the 1983 Mercury Task Force. Spreadsheets documenting the data and calculations used to quantify mercury releases to air are provided in Appendix H. Section 4.5 discusses the data and methodology used to estimate releases to water. Spreadsheets documenting the data and calculations used to quantify mercury releases to water are provided in Appendix I.

Table 4-3: Sources of Historical Data on Mercury Concentrations in Liquid Effluent and Water Flow Rates

EFPC Water Concentration Data

- **Y-12 Plant Quarterly Reports** (ChemRisk Repository No. 3264). These reports present quarterly-average EFPC concentrations for 1953-62. Data presented in these reports were compared to quarterly-averages presented in the 1983 Mercury Task Force Report.
- **Y-12 Technical Division Monthly Progress Reports** (ChemRisk Repository No. 3264). These reports present monthly concentrations measured at the Y-12 site boundary that are referred to in the quarterly reports, for 1954-58.
- **Y-12 Monthly Surface Water Sampling Reports** (ChemRisk Repository No. 3259). These reports present monthly-average EFPC concentrations for 1955-64.
- **1982 Memo from Fee (Y-12 Plant Manager) to Hickman (ORO)** (Fee and Sanders 1982) (ChemRisk Repository No. 3259). This memo includes tables prepared by M. Sanders of weekly concentrations in EFPC for 1955-82.
- **The 1983 Mercury Task Force Report** (ChemRisk Repository No. 3272). This report presents quarterly estimates of pounds of mercury released to EFPC and quarterly-average EFPC concentrations for 1955-82 (UCCND 1983a). The supporting documentation for the water release data is Fee and Sanders (1982).
- **ORR Environmental Monitoring Reports** (ChemRisk Repository No. 3259). These reports present annual-average EFPC concentrations for 1981-93.

Water Flow Rate Data

- **Water Flow in EFPC, May 1955 - October 1956** (ChemRisk Repository No. 3260). This document includes typed records of daily measurements for 1955-56.
- **The 1983 Mercury Task Force Report** (ChemRisk Repository No. 3272). This report presents quarterly-average stream flow rates for 1955-82 (UCCND 1983a).
- **Y-12 Monthly Surface Water Sampling Reports** (ChemRisk Repository No. 3259). These reports present monthly-average flow rates for 1956-64.
- **Y-12 Plant Quarterly Reports** (ChemRisk Repository No. 3264). These reports present quarterly-average flow rates for 1956-72.
- **Original recording charts and handwritten records of daily stream flow measurements** (ChemRisk Repository No. 3260). These records, provided by the Y-12 HSEA Division Surface Water Group, include daily measurements for 1972-85. Averages were calculated by the Task 2 team.
- **USGS daily measurements** (ChemRisk Repository No. 3260). These records, provided as computerized spreadsheet files by the Y-12 HSEA Surface Water Group, include daily measurements for 1985-93. Averages were calculated by the Task 2 team.

The total mercury inventory used in Building 9201-2 operations was 321,753 pounds (Kite 1957). When the Colex pilot plant was shut down in 1955, 186,596 pounds of mercury were transferred to Buildings 9201-5 and 9201-4 (Kite 1958), leaving 135,157 pounds of mercury unaccounted for. It is known that mercury remains in the Building 9201-2 structure. For example, small amounts of mercury were recovered (e.g., 800 pounds from a pipe) during closure operations in 1983, and when the first floor was converted to office space in 1983, mercury seeped out of several walls and small beads of mercury were visible in the basement (Turner 1990).

Historical information on mercury releases to air and water and spills to the ground from the Colex pilot plant (Building 9201-2) are summarized below.

Airborne Mercury Releases from the Colex Pilot Plant

According to the 1983 Mercury Task Force Report, the Colex pilot plant was built in available building space in Building 9201-2 and only minimum modifications were made to the building. As a result, ventilation and spill containment were less than desirable and resulted in high indoor air concentrations of mercury (UCCND 1983a). Building air monitoring was conducted during 1953 through 1955, and continued after operations were shut down until 1959, likely because of basement contamination.

The project team located Building 9201-2 air data for 1953 through 1959 in weekly and monthly solvent air analysis reports from the Industrial Hygiene Section (ChemRisk Repository No. 3262) and in a 1952 building ventilation study (Baumann 1952). Three Y-12 memoranda (Postma 1971; Edwards 1972; Landis 1976) discuss results of periodic mercury vapor monitoring in the Building 9201-2 basement in the 1970s, and the recommendation that no further action be taken to decontaminate the building. The highest mercury air concentrations measured in the three surveys of the basement were 0.08 mg m^{-3} (1971), 0.04 mg m^{-3} (1972) and 0.03 mg m^{-3} (1976).

Losses of mercury from Building 9201-2 were not estimated in the 1983 Mercury Task Force Report. Limited ventilation in Building 9201-2 suggests that air releases to the environment were negligible compared to releases from production scale lithium separation operations. The project team estimated Building 9201-2 releases for 1953 to 1957 using ventilation rates estimated by Choat (see Appendix G), but did not estimate mercury releases to air for 1958 and 1959 because the basement exhaust was inoperable. The project team's best estimate of mercury releases to air from Building 9201-2 between 1953 and 1957 was 598 pounds. This estimate is less than 1% of the total mercury released to air from Y-12.

Waterborne Mercury Releases from the Colex Pilot Plant

Several inches of water was used to cover the floor during operations in the 9201-2 Colex pilot plant to reduce airborne mercury concentrations. This mercury contaminated water was discharged into EFPC. Mercury in the contaminated water would have been reflected in water concentrations measured in EFPC during this time period (see Section 4.5).

Spills in the Colex Pilot Plant

The project team reviewed information on three spills of mercury in Building 9201-2 between 1952-1955. These were (1) rupture of a check valve during a pump test, (2) splitting of a pipe seam during a test, and (3) a pump failure. Efforts were made to recover the mercury at the time of each spill, but most of it ran deeper into the ground during the excavation operation and was not recovered. The total loss estimated from the three spills is 95,000 pounds. Dirt from the excavations was later processed at the Building 81-10 mercury recovery facility. However, mercury recovered at the 81-10 facility could not specifically be attributed to this spill, since the dirt was not segregated from dirt from other spills. Following the spills, a heavy layer of sulfur was added to the surface of the basement's dirt floor to contain mercury vapor resulting from subsurface mercury (UCCND 1983a; Turner 1990).

The project team did not develop a source term for mercury spills to soil because any mercury runoff to EFPC within the plant boundary and before the sampling location would have been included in the mercury concentrations measured at the site boundary.

4.3.2 Releases from the Elex Production Scale Facility (Building 9204-4)

The criteria for an Elex production plant to be built in Building 9204-4 (also known as Beta-4) were fixed in December 1951 due to the urgency of the lithium isotope production project (ADP History 1948-51). Criteria were based on the Elex pilot plant tests made in 1950 and 1951 in Building 9201-2 (see Appendix A). Construction of the Elex production plant was completed in about 17 months (Sapirie 1956). Flasks of mercury were delivered to Building 9204-4 between December 1952 and January 1953 and Union Carbide employees emptied the flasks into the lithium processing equipment. The total mercury going into the building was weighed and recorded as 1.5 million pounds (UCCND 1983a; Center 1953).

On August 18, 1953 the first half of the Elex production plant was placed on line (ADP Chronology 1950-54), and on August 24, 1953, the first product was withdrawn from the system (Sapirie 1956). The Elex production plant included two cascades that were serviced by a common group of auxiliary systems (e.g., evaporator, demineralizer, lithium feed purification). The two cascade areas were referred to as the North Plant and the South Plant. During its operation, the facility produced three times its design capacity (UCNC 1957; Sapirie 1956). Hydrogen gas produced during lithium separation was vented to the outside air to reduce the risk of explosion**S** this gas contained mercury vapor.

The Elex production plant was shut down on March 16, 1956. A March 21, 1956 letter from S. R. Sapirie, Manager of USDOE Oak Ridge Operations, to C.E. Center, Vice-President, Union Carbide, states:

> "We have been advised that the General Manager has authorized the temporary shutdown of the 9204-4 Plant as a means of accelerating the cleanup of the mercury health problem in the ADP alpha plants [9201-5 and 9201-4]. This authorization was granted with the understanding that the 9204-4 Plant would be in a ready standby condition requiring only three months to be back in full operation." (Sapirie 1956).

In September 1956, the decision was made to strip the process equipment from Building 9204-4. Stripping of the building was completed by December 1, 1956. All Building 9204-4 process equipment was opened and cleaned, and all process solutions and 1,417,000 pounds of mercury were directly transferred by pipeline to the two Colex production buildings (9201-4 and 9201-5; UCCND 1983a). The major pieces of equipment were stored in the Y-12 salvage yard and not sold as surplus property or scrap. In 1958 and 1959, two furnaces were installed and operated west of Building 9720-26 by two local scrap dealers to process the scrap metal for sale. Scrap copper, steel, stainless steel, and nickel were melted and cast into ingots. The furnaces were disassembled and removed sometime after 1959. In addition, according to Alloy Division Solvent Air monthly reports (LaFrance 1955-60), beta extract operations were conducted in Building 9204-4 in 1958 and 1959.

Mercury releases from the Elex production plant in Building 9204-4 are described below. Identified releases were via airborne emission pathways.

Airborne Mercury Releases from the Elex Production Plant

The 1983 Mercury Task Force Report estimated air losses of mercury from Building 9204-4 arising from mercury emissions in the hydrogen off-gas and from outdoor smelting of mercury-contaminated scrap metal. Although the Building 9204-4 Elex production plant used only 6% by volume of the mercury used in Colex, had few leaks and spills, and operated for a shorter time period than Colex, daily mercury air releases from the Elex processes were high, approaching the quantities later released routinely by the much larger Colex buildings. This was because the hydrogen gas was not scrubbed or otherwise treated to reduce mercury content (UCCND 1983a). An October 1953 study of mercury in Building 9204-4 stack exhaust indicated that 8.46 lbs of mercury were exhausted from the stack per day (Baumann 1953a). In contrast, the total mercury loss in hydrogen off-gas during Colex operations in Buildings 9201-5 and 9201-4 was only 10 pounds, due to the effectiveness of Colex's water-bath scrubber system in removing mercury from the gas. Based on the 1953 stack exhaust study, the 1983 Mercury Task Force estimated that a total of 8,300 pounds of mercury were exhausted from Building 9204-4.

Theoretically, the building exhaust would have included mercury from the building air and from the hydrogen off-gas. The Task 2 project team compared the 1953 stack exhaust study to estimates of mercury releases calculated using building air data and 1983 Mercury Task Force estimates of mercury in hydrogen off-gas. The following were the primary sources of building air concentration data used by the project team to calculate quarterly average air concentrations in Building 9204-4 for the specified time periods:

- 1953 (3rd quarter) to 1954 (4th quarter)**S** Weekly, monthly and quarterly Health Physics reports
- 1955 (1st quarter) to 1956 (3rd quarter)**S** Monthly Industrial Hygiene solvent air analysis reports *(Elex was shutdown in March 1956, and stripped during 4th quarter 1956)*

- 1958 (2nd quarter) to 1959 (4th quarter)**S** Monthly Industrial Hygiene solvent air analysis reports *(Beta extract operations occurred during 1958 and 1959)*

The comparison showed the amount of mercury in the building exhaust was greater than in the other two sources combined. Consequently, for the period of lithium separation operations (1953-1st quarter 1956), the Task 2 project team used the results of the stack exhaust study to estimate mercury releases to air from Building 9204-4. For 1956 building stripping operations and 1958-59 beta-extract operations, the project team used building air concentrations and Choat's estimates of building ventilation rates (Choat 1996; see Appendix G) to estimate mercury releases from the building, since hydrogen off-gas was not being generated.

No pollution control equipment was installed on the furnaces used in 1958 and 1959 to smelt scrap metal from Building 9204-4. The 1983 Mercury Task Force estimated that a total of 5,000 pounds of mercury were lost to the air during the smelting operations (UCCND 1983a). The project team did not locate any supporting information that could be used to revise this estimate, and therefore included the Mercury Task Force estimate of 5,000 lbs. in their estimate of mercury released to air, assuming that 2,500 lbs. were lost to air in 1958 and 2,500 lbs in 1959.

The project team's best estimate of mercury released to air from Building 9204-4 and the smelting operations was 17,070 pounds (compared to the 1983 Mercury Task Force estimate of 13,300 pounds). This estimate is about 23% of the total mercury estimated by the Task 2 team to have been released to air from Y-12.

4.3.3 Releases from Colex Production Scale Facilities (Buildings 9201-5 and 9201-4)

Two Colex production plants were constructed in Y-12 Buildings 9201-5 and 9201-4 between 1954 and 1955. The buildings were originally built to house electromagnetic enrichment operations for uranium during World War II. Building 9201-5 (known as Alpha-5) was authorized for the Colex production plant in December 1953 and began operations on January 20, 1955 with six separation cascades. Building 9201-4 (known as Alpha-4) was authorized in June 1954 and began operations in June 1955, with four cascades of larger diameter than the six cascades in 9201-5 (UCNC 1957). Photograph 2 shows Buildings 9201-5 and 9201-4 as they appeared in 1954.

Approximately 24 million pounds of mercury were used in the Y-12 Colex production operations. In 1955, the Rust Engineering Company constructed a special mercury unloading facility just north of Building 9201-5. Flasks of mercury were unloaded from trucks at three docks by Rust personnel. The flasks were held in the storage yard just west of Building 9204-4 and emptied into a trough that filled a three-section storage tank. The transfer station was an open structure, and large fans were used to ensure adequate ventilation. Rust employees did not weigh the mercury that was delivered. Transfer vouchers for the mercury were prepared from General Services Administration invoices after the deliveries, assuming 76 pounds were received per flask (UCCND 1983a). Photographs 3 and 4 show the mercury flask storage area and the flasks being emptied for use in the Colex buildings in 1955. Photograph 5 is a photograph of

the large propeller-type exhaust fans (6-ft in diameter) installed in 1955 to supply additional exhaust ventilation to the Colex buildings.

The 9201-5 facility was shut down in March 1959, but restarted for six months in December 1962 to produce marble (^{7}Li). The Building 9201-4 facility was shut down and put into standby in December 1962 (Sapirie 1962). An October 3, 1962 letter from S. R. Sapirie, Manager, Oak Ridge Operations, to C.E. Larson, Vice-President, Union Carbide, states:

> "The [Atomic Energy] Commission has decided to discontinue the production of lithium-6 [in Building 9201-4] because the present inventory and scheduled returns would provide the weapons requirement of lithium-6 for a period of approximately three years."

The letter further states that the 9201-4 facility was to be placed in a standby condition such that production could be resumed upon six months notice, and that no process equipment was to be removed from the building (Sapirie 1962). The 9201-4 mercury inventory was bottled in 1977. It is estimated that 450,000 pounds of mercury may remain in Building 9201-4 process equipment today (UCCND 1983a; Oak Ridger 1992).

Building 9201-5 was stripped from 1965 to 1967. A June 4, 1965 memorandum from D.A. Jennings, Union Carbide, to J. W. Ebert, Union Carbide, states that stripping of Building 9201-5 started on March 29, 1965, and that 30 months would be required to completely strip the building (Jennings 1965). Two April 1966 memoranda refer to a search for a mercury deposit in a dirt excavation in a Building 9201-5 fan room, and an extension of the completion date for Building 9201-5 stripping from September 30, 1966 to June 30, 1967 (Smith 1966; Hibbs 1966). During the stripping operations, approximately 50,000 pounds of mercury were lost in a March 1966 spill (discussed below; UCCND 1983a; USAEC 1966). According to the 1983 Mercury Task Force Report, 262,000 pounds of mercury were recovered from disassembled Building 9201-5 process equipment, and an additional 54,000 pounds were reportedly recovered by scrap dealers who purchased the scrap metal and processed it off-site. These quantities are in addition to the majority of the building's mercury inventory, which was drained and rebottled prior to equipment disassembly.

The project team and members of the Oak Ridge Health Agreement Steering Panel were given a tour of the Alpha-4 Colex production building by Mr. O.K. Clotfelter. Photograph 6 shows the amalgam maker trays located on the third floor of the Alpha-5 building taken during that tour.

Mercury releases from the Colex production plants (Building 9201-5 and 9201-4) are described below. Identified releases were via airborne emission pathways and spills to the ground.

Airborne Releases from Colex Production Plants

Several sources of building air data collected in the two Colex process buildings for the period from 1955 to 1962 were identified by the Task 2 team. Building ventilation drawings for both buildings and an exhaust ventilation study for 9201-5 were also located.

The following records were the primary sources of building air concentration data used by the project team to calculate quarterly average Building 9201-5 and 9201-4 air concentrations for the specified time periods:

- 1955 (1st quarter) to 1957 (1st quarter)**S** LaFrance (1957) (Buildings 9201-5 and 9201-4)
- 1955 (2nd quarter) to 1960 (1st quarter)**S** Monthly Solvent Air Analysis Reports (Buildings 9201-5 and 9201-4)
- 1955 to 1961**S** Y-12 Plant Quarterly Reports (Buildings 9201-5 and 9201-4)
- 1960 (1st quarter) to 1962 (4th quarter)**S** 1983 Mercury Task Force Report (Building 9201-4)

Daily building air concentration measurements were originally recorded on Solvent Air Analysis Report (SAAR) forms. The project team located SAAR forms and monthly ledger averaging sheets for Y-12 lithium separation buildings for 1952 to 1963 in boxes at the Y-12 Records Center. Some daily averaging calculations were recorded directly on SAAR forms in red pencil, and some monthly averaging calculations were recorded on handwritten ledger sheets located in the boxes with the SAAR forms. The project team spot-checked concentrations recorded on the SAAR forms against average building air concentrations reported in the above sources. The calculated averages were correct for all buildings and time periods reviewed.

For 1957-61, there is good agreement between all sources of building air concentration data. However, for 1955-56, the concentrations reported in the Y-12 Quarterly Reports are slightly lower than the other three sources. For these two years, the project team used the higher reported concentrations.

In late 1955, Y-12 Engineering conducted a study of mercury concentrations in air exhausted from Building 9201-5 (Little 1956). The first estimates of mercury losses from Building 9201-4 were made in 1977**S** these estimates were based on the Little (1956) study of Building 9201-5 air exhaust because it was assumed that the two buildings were similar. The 1983 Mercury Task Force Report revised estimates of mercury losses through ventilation from Buildings 9201-5 and 9201-4, using quarterly-average building air concentrations for each building. However, the 1983 Mercury Task Force continued to use exhaust flows for Building 9201-5 to estimate mercury losses from Building 9201-4, since a ventilation study similar to Little's (1956) had not been conducted for Building 9201-4.

In a reanalysis of Building 9201-5 and 9201-4 ventilation rates conducted for the project team, Choat found that the 1983 Mercury Task Force assumption of equal exhaust air flow in 9201-5 and 9201-4 was incorrect, and that the air flow in 9201-4 was actually twice the air flow in 9201-5, based on design ventilation drawings (Choat 1996) (see Appendix G). The project team used Choat's estimates of building ventilation rates, and quarterly average building air concentrations, to estimate mercury releases to air from Buildings 9201-5 and 9201-4. Using these data, the project team estimated that approximately 19,900 pounds of mercury were released from Building 9201-5 and 33,300 pounds were released from Building 9201-4 (compared to 1983 Mercury Task Force estimates of 19,500 pounds and 18,400 pounds, respectively). These releases represent 46% and 27%, respectively, of the total mercury estimated by the project team to have been released to air from Y-12.

During 1965 to 1967 stripping operations, air concentrations were measured in Building 9201-5. An April 1966 memorandum states, "mercury vapor readings in the immediate stripping area are frequently above maximum allowable limits (0.1 mg m^{-3}), and respirators are [therefore] required" (Hibbs 1966). No information on ventilation conditions or additional air concentration data during stripping operations were located by the project team. It is likely that air emissions of mercury were significantly lower than during production operations, because the building ventilation system would not be operating as it did during production operations (personal communication with E.E. Choat, formerly of Y-12 Engineering). Therefore, air emissions from the stripping operations were not estimated or included in the project team's estimate of mercury released to air from Building 9201-5.

Spills in the Colex Production Plants

Five spills totaling 285,500 to 500,000 pounds of mercury occurred during production operations in Buildings 9201-5 and 9201-4. These are summarized below:

- On January 1, 1956, a coupling on the discharge side of a pump broke. It took 3-4 minutes to shut down the cascade. 113,000-170,000 pounds of mercury were estimated to have been spilled. Visible mercury was recovered in the building (43,000-100,000 pounds estimated), and 70,000 pounds were estimated to have been spilled to the ground through the fan room floor (UCCND 1983a).

- On July 17, 1956, an operator error in valving occurred while transferring mercury from Building 9201-5 to Building 9201-4, on the ramp area north of Building 9201-5. It was estimated that 22,500-90,000 pounds of mercury were spilled. Visible mercury was recovered by shoveling (estimated 5,000 pounds), and it was estimated that a maximum of 85,000 pounds of mercury were not recovered. The dirt was later processed at the Building 81-10 mercury recovery facility; however, any mercury that was recovered could not be specifically attributed to this spill, since dirt from different spills was not segregated (UCCND 1983a).

- In mid-1956, an operator error in valving occurred during the transfer of mercury from Building 9204-4 to Building 9201-5, at the mercury dumping station. It was estimated that 22,500-90,000 pounds of mercury was spilled on the ground at the mercury transfer station. Visible mercury was recovered by shoveling (estimated 5,000 pounds), and it was estimated that a maximum of 85,000 pounds of mercury were not recovered. The dirt was later processed at the Building 81-10 mercury recovery facility; however, any mercury that was recovered could not be specifically attributed to this spill (UCCND 1983a).

- On November 15, 1956, a cascade plugged and caused an overflow in the northwest corner of Building 9201-5. It took 3-4 minutes to shut down the cascade. 22,500-45,000 pounds of mercury were estimated by volume to have been spilled. Visible mercury was recovered by shoveling (estimated 5,000 pounds), and it was estimated that a maximum of 40,000 pounds of mercury were not recovered. The dirt was later processed at the Building 81-10 mercury recovery facility; however, any mercury that was recovered could not be specifically attributed to this spill (UCCND 1983a).

- On March 28, 1966, a leak occurred in a sight glass in a mercury collection tank in Building 9201-5. 105,000 pounds of mercury were estimated by volume to have been spilled. This was the only spill quantified by measurement. 55,000 pounds of mercury were recovered, and the remainder seeped through construction expansion joints to the ground under the concrete floor. It was determined by measurement that 49,853 pounds of mercury were not recovered (UCCND 1983a; USAEC 1966).

The project team did not develop a source term for mercury spills to soil because any mercury runoff to EFPC within the plant boundary and before the sampling location would have been included in the mercury concentrations measured at the site boundary (see Section 4.5). Further, some of the spilled mercury not recovered at the time of each spill was recovered at the on-site mercury recovery facility from dirt excavated at the spill sites (3.6 million pounds of mercury were recovered at the mercury recovery facility during its operation). The project team estimated mercury released to air from the mercury recovery facility (see Section 4.3.4).

4.3.4 Releases from the Mercury Recovery Facility (Building 81-10)

A mercury recovery facility was operated at Building 81-10 from March 1957 to September 1962. The facility was constructed to recover mercury by physical separation (draining or decanting) or by distillation in a furnace and recovery by condensation (Napier 1975). Mercury recovered at Building 81-10 was returned to the operating inventory in the Colex process buildings (UCCND 1983a).

The mercury recovery facility, an outdoor installation, was comprised of a storage shed (approximately 40 feet wide by 100 feet long) and a processing area (approximately 30 feet wide by 50 feet long). Photograph 7 shows the facility just before it was torn down in November 1983. The furnace was located inside the scaffolding shown in the photo. Drums of material to be processed in the furnace were stored under the adjacent shed. The furnace processed evaporation filter sludge, cascade decomposer graphite, various filter solids, sump and tank cleanings, and other waste materials from the Colex process. The furnace off-gas was passed through a condenser to recover mercury. A forced-air fan provided air to the burners for combustion with the fuel gas (UCCND 1983a). Water was used to cool the mercury as it fell from the condenser and as a scrubbing agent in the scrubber (UCCND 1983a). The furnace is known to have produced high mercury air concentrations in the immediate vicinity of the building (UCCND 1983a). Operation of the furnace was discontinued in July 1962, while physical separation operations continued through September 1962 (Log sheets 1957-62).

Mercury releases to air, water, and spills to the ground from the mercury recovery operations (Building 81-10) are described below.

Airborne Mercury Releases from Mercury Recovery Operations

A number of problems were immediately apparent upon the initial operation of the 81-10 furnace. The most important of these problems was the inadequacy of the off-gas system, which allowed pressurization of the furnace and resulted in mercury contamination of the area. A Y-12 employee stated that the mercury vapor concentration in the air surrounding this outside installation "pegged his mercury vapor meter at least once" (UCCND 1983a). The furnace was shut down for repair and modification in May or June of 1957 (Morehead 1957). Following this shutdown, airborne mercury was sampled and analyzed, and losses were said to be "insignificant" (UCCND 1983a).

The project team located daily log sheets of mercury air measurements taken at various 81-10 locations from 1957 to 1963 in boxes belonging to the Radiation Safety Department at the Y-12 Records Center. These data were not used in the estimate of mercury released to air at 81-10, due to the identification of a furnace efficiency study (Reece 1959) and log sheets quantifying mercury recovered at the 81-10 facility (Y/HG-0023; Y/HG-0005).

The project team used logs of the quantities of mercury recovered at the 81-10 facility (about 3.6 million pounds) and the 1959 furnace efficiency study (Reece 1959, conducted after the 1957 improvements were made to the furnace) to estimate mercury released to air as a result of mercury recovery operations. Air emissions from the roasting furnace were estimated based on a test run from May 4, 1959 to May 12, 1959 (Reece 1959). The total recovery of mercury was 341 pounds, and the total recovery plus known losses was 371 pounds, indicating a recovery efficiency of 341/371 = 92%. Therefore, recovery should be about 92% of furnace input. The loss to stack gases during the test run was 0.18 pounds. This was 0.0005 (0.05%) of the total recovery plus known losses (0.18/371 = 0.0005). The remaining 29.8 pounds (7.95%) of known mercury losses were measured in waste products from furnace operations, including furnace ash, fly ash, and scrubber water. On this basis, the annual air emissions from the mercury roasting

furnace were estimated to be 0.05% of the input to the furnace. Stack losses prior to the May 1957 improvements were measured during April and May. Average stack losses during these two sampling periods [average 0.023 lb d^{-1} (range 0.013 - 0.037 lb d^{-1}, N = 3) and 0.035 lb d^{-1} (range 0.005 - 0.069 lb d^{-1}, N=6), respectively] were similar to stack losses measured after the improvements during two days in June 1957 [average 0.037 lb d^{-1} (range 0.016 - 0.060 lb d^{-1}, N = 6) and 0.045 lb d^{-1} (range 0.038 - 0.057 lb d^{-1}, N=6), respectively]. The efficiency study was therefore considered to be representative of stack losses both before and after furnace improvements.

The decanting process is described as draining the elemental mercury that settled at the bottom of the feed hopper into drums prior to roasting the remaining feed material in the furnace for further mercury recovery (Napier 1995). Volatilization of mercury from the decanting operation was assumed to be negligible compared to emissions of mercury vapor from the furnace.

Based on the assumption that annual air emissions from the mercury roasting furnace were 0.05% of the estimated input to the furnace, the estimated total mercury release to air between 1957 and 1962 was approximately 930 pounds (see Appendix H). This estimate was less than 2% of the total mercury estimated by the project team to have been released to air from Y-12. Losses of mercury to air from Building 81-10 were not estimated in the 1983 Mercury Task Force Report.

Waterborne Mercury Releases from Mercury Recovery Operations

When originally installed, the mercury recovery system in Building 81-10 had a containment system for spilled mercury and mercury-contaminated sludges, which flowed into a sump. The effluent from the sump flowed into a nearby ditch and ultimately into EFPC. In 1958, a large secondary sump was installed across the road. Releases of mercury to EFPC from the sump effluent would be included in the mercury monitoring data collected at the Y-12 Plant boundary (see Section 4.5).

Spills at the Mercury Recovery Facility

Some mercury leaked or was spilled from containers on the mercury recovery facility storage pad and was dispersed into the adjoining strip of ground. In August 1971, there were some core drillings made to determine the extent of mercury in the ground around the facility (Guettner 1971). The core samples taken in 1971 were used to develop the estimate in the Mercury Task Force Report of 3,000 pounds of mercury lost to the ground at Building 81-10 (UCCND 1983a).

The project team did not develop a source term for mercury spills to soil because any mercury runoff to EFPC within the plant boundary and before the sampling location would have been included in the mercury concentrations measured at the site boundary.

4.3.5 Releases from the Y-12 Steam Plants (Buildings 9401-1, 9401-2, 9401-3)

Construction of a new 34,000 ft^2 steam plant (Building 9401-3) to meet the increased electricity requirements of the Alpha-5 and Alpha-4 lithium separation operations began on January 19, 1954 and was completed in June 1954. The two old 12,000 ft^2 steam plants, Buildings 9401-1 and 9401-2, were subsequently dismantled, and the buildings used for other purposes (Thomason and Associates 1996). On the basis of gas and coal costs at the time, the decision to use pulverized coal as a primary fuel in the new steam plant was made on December 10, 1953 (author and date unknown- Chronology of Alpha-5 Plant 1953-54). A January 27, 1956 letter to R. C. Armstrong, AEC Director, from J. P. Murray, Y-12 Plant Manager, presented cost estimates associated with the conversion of two of the four boilers in the 9401-3 steam plant for burning natural gas instead of pulverized coal during the six summer months of the year (Murray 1956). The April-June 1956 Y-12 Plant Quarterly Report states that this conversion to natural gas during the summer months was 80% complete. The steam plants did not have any controls to prevent or reduce mercury emissions in the 1950s. Photograph 8 shows one of the Y-12 steam plants, 9401-2.

Mercury releases from the Y-12 steam plants are described below. Identified releases were via airborne emissions pathways.

Airborne Mercury Releases from the Y-12 Steam Plants

In a report titled "Mercury in Ambient Air at the Oak Ridge Y-12 Plant, 1986-1990" (Turner et al. 1991), the establishment of an ambient air monitoring program for mercury at Y-12 is described. The report states that burning of coal at the Y-12 steam plant (Building 9401-3) contributes to above-background concentrations of mercury in air at the Y-12 site. According to the authors, prior to 1989, the 9401-3 steam plant burned 70,000 kg of coal per year, containing 0.5 mg mercury per kg of coal. This would result in an annual mercury release to air of approximately 0.077 pounds. However, Y-12 Plant Quarterly Reports state that a total of 3.6×10^8 kg of coal was burned at Y-12 between 1956 and 1959. Estimates of coal burned at the Y-12 steam plants provided in the Y-12 Quarterly Reports for 1956 through 1959 were used in the project team's estimate of mercury released from coal burning, instead of the estimate provided in Turner et al. (1991) for 1989 and earlier. The annual releases of mercury to air from coal burned at the 9401-3 steam plant are presented below assuming 0.5 mg mercury per kg of coal.

Table 4-4: Estimated Annual Releases of Mercury to Air from Coal Burned at the 9401-3 Steam Plant

Year	Coal Consumption ($\times 10^8$ kg)	Estimated Mercury Emissions (lb)
1956	1.74	192
1957	0.74	82
1958	0.51	56
1959	0.625	69

No information on the annual tonnages or mercury content of coal burned in the two old steam plants (Buildings 9401-1 and 9401-2) prior to 1956, or air concentrations of mercury prior to 1986, were located by the project team during the document search. For the years 1953-55, the project team assumed that 192 pounds of mercury per year were emitted from the two old steam plants, since 9401-3 was reported to have the same capacity as 9401-1 and 9401-2 combined. For 1960-62, it was assumed that 69 pounds of mercury per year were emitted from the new steam plant (9401-3), since the plant began burning natural gas instead of coal in 1959, primarily during the summer months.

Based on these calculations, the project team estimated that approximately 1,182 pounds of mercury were released to air from the Y-12 steam plants. This estimate is less than 1.5% of the total mercury estimated by the project team to have been released to air from Y-12. Losses of mercury from ORR steam plants were not estimated in the 1983 Mercury Task Force Report.

4.4 Estimates of Mercury Releases to Air

This section summarizes the project team's estimates of airborne releases of mercury from the Y-12 Plant and discusses the uncertainty in building air concentrations and ventilation rates. In addition, this section summarizes information describing the speciation of mercury released to air. Documentation of instrument development and calibration, data on measurement of mercury standards, and descriptions of the airborne mercury monitoring program (presented in Section 3.3 and Appendix B), were used to evaluate the quality of Y-12 air monitoring data for estimating releases of mercury to air from Y-12 buildings and to develop uncertainty factors for application to air release estimates.

4.4.1 Summary of Estimated Mercury Releases to Air

The project team's best estimate of the total mass of mercury released to the ambient air during lithium isotope separation operations is 73,000 pounds. By comparison, the 1983 Mercury Task Force estimate was 51,000 pounds. This is an increase of 22,000 pounds or 43%. Table 4-5 summarizes the estimated annual releases of mercury to air for each facility or process.

Table 4-5: Task 2 Estimates of Annual Mercury Releases to Air from Y-12 Facilities (lb)

Year	Bldg 9201-2	Bldg 9201-4	Bldg 9201-5	Bldg 9204-4	Bldg 81-10	Steam Plants	Total
1953	162	0	0	1,142	0	192	1,496
1954	200	0	0	3,046	0	192	3,438
1955	115	9,280	9,212	3,807	0	192	22,606
1956	79	6,012	5,848	1,700	0	192	13,831
1957	42	3,486	2,077	0	215	82	5,902
1958	0	3,466	1,381	3,959	381	56	9,243
1959	0	3,286	912	3,416	120	69	7,803
1960	0	3,085	492	0	68	69	3,714
1961	0	2,324	0	0	82	69	2,475
1962	0	2,324	0	0	63	69	2,456
TOTALS	**598**	**33,263**	**19,922**	**17,070**	**929**	**1,182**	**72,964**
(To two significant figures)	**(600)**	**(33,000)**	**(20,000)**	**(17,000)**	**(930)**	**(1,200)**	**(73,000)**

Table 4-6 summarizes differences between the estimates by the Task 2 team and the 1983 Mercury Task Force.

Table 4-6: Comparison of Mercury Task Force and Dose Reconstruction Team Estimates of Mercury Released to Air

Building	1983 Task Force Estimate (lb)	Task 2 Estimate (lb)	Difference (lb)	Difference (%)
9201-5	19,473	19,922	+449	+2
9201-4	18,447	33,264	+14,817	+80
9204-4	13,300	17,070	+3,770	+28
9201-2	0	598	+598	—
81-10 Mercury Recovery	0	929	+929	—
Steam Plants	0	1,182	+1,182	—
TOTALS (To 2 significant figures)	**51,220 (51,000)**	**72,965 (73,000)**	**+21,745 (+22,000)**	**+43**

Figure 4-3 allows comparison of 1983 and 1996 estimates of mercury released to air.

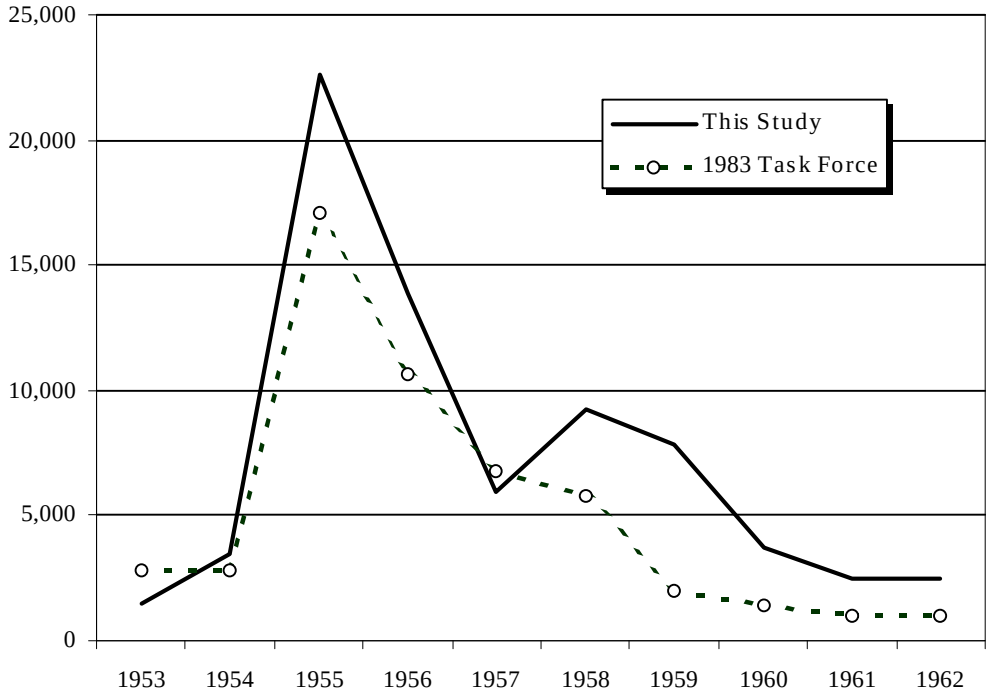


Figure 4-3: Annual Airborne Mercury Release Estimates by the Task 2 Project Team and the 1983 Mercury Task Force (pounds)

4.4.2 Uncertainty in Measurements of Concentration and Air Flow Rate

The project team estimated that the uncertainty in the reported mercury air concentrations determined by Y-12 portable mercury vapor detectors is ± 40%. This estimate is based on a lack of data from standards and duplicate measurements (Presbo 1996). Under current sampling and analysis quality assurance guidelines, data of this type are typically collected and analyzed concurrently with environmental samples to assess the precision and accuracy of the sampling and analytical methods.

Choat (1996) estimated that the uncertainty in building exhaust air flow rates for Buildings 9201-5, 9201-4, and 9204-4 is ± 3%. These estimates of exhaust air flow are based on actual ventilation design drawings; however, it is reasonable to assume there were minor variations in as-built conditions compared to design conditions, and minor alterations were made for spot ventilation in problem areas (Choat 1996). Choat (1996) estimated a higher uncertainty in the exhaust air flow rate for Building 9201-2, ± 50%, since there were no drawings of the 9201-2 building ventilation system except for the specific equipment used in the pilot plant operations, (personal communication with E. Choat).

4.4.3 Documentation of Calculations and Numerical Results

Appendix H presents the tables and spreadsheets used by the project team to document the source term calculations for mercury released to air. Tables and spreadsheets in Appendix H are as follows:

- Table H-1 summarizes the air concentration and flow rate data used to estimate releases of mercury to ambient air for each year from 1953-62, and the uncertainty associated with these concentrations and flow rates.

- Table H-2 summarizes mercury releases to the air from Y-12 lithium separation buildings and steam plants between 1953 and 1962.

- Tables H-3 through H-7 present the calculations of mercury releases to air from Buildings 9201-5, 9201-4, 9204-4, 9201-2, and 81-10 based on building air concentrations and exhaust flow rates, incorporating the revised estimate of exhaust flow rate for Building 9201-4.

- Table H-8 compare monthly and quarterly building air mercury concentrations from four sources of data for Buildings 9201-5, 9201-4, 9204-4 and 9201-2.

- Table H-9 shows calculations used by the project team to check the 1983 Mercury Task Force Report calculation of pounds of mercury released to air.

4.4.4 Mercury Speciation in Y-12 Air Releases

Elemental mercury was used in the lithium isotope separations process. Because of its relatively high vapor pressure, elemental mercury is relatively volatile compared to most metals and airborne mercury present in the lithium isotope separations work areas likely consisted largely of elemental mercury vapor. Per the 1983 Mercury Task Force, in-plant worker exposures to mercury were almost entirely to the metal vapor (UCCND 1983a).

4.5 Estimates of Mercury Releases to Surface Water

This section discusses the data and methodology used by the project team to estimate releases of mercury from the Y-12 Plant to EFPC and discusses the uncertainty in the EFPC water concentrations. In addition, this section summarizes information describing the speciation of mercury released to water.

Information from historical documents describing method development, instrument calibration, measurement of mercury standards, and the EFPC mercury monitoring program, summarized in Section 3.4 and Appendix B, was also used to evaluate the quality of historical monitoring data for estimating releases of mercury to EFPC and to develop uncertainty factors for application to water release estimates.

4.5.1 Calculation of Mercury Releases to EFPC

In general, the project team calculated annual mercury releases to EFPC using the lowest level of summary data available (for example, where available, release estimates were based on individual weekly composite sample measurements, rather than monthly or quarterly averages). The project team located only a few of the original analytical cards submitted with the water samples. Therefore, the team relied primarily on summaries presented in monthly and quarterly Y-12 reports and ORR Site Environmental Reports. In the absence of data from other sources, tabulated data presented in the 1983 Mercury Task Force Report were used.

Concentration Data

From 1953 to 1973, samples collected in EFPC were composited for weekly analysis for mercury. After 1973, samples were composited for monthly analysis. The following records were the primary sources of water concentration data used by the project team to calculate mercury releases to EFPC for the specified time periods:

- 1953 to 1955**S** In-plant memoranda
- 1955 (2nd quarter) to 1964 (2nd quarter)**S** Monthly Surface Water Sampling Reports
- 1964 (3rd quarter) to 1982 (3rd quarter)**S** Fee and Sanders (1982)
- 1982 to 1993**S** ORR Site Environmental Reports

Individual results from weekly composite samples were reported in Y-12 memoranda for 1953 to 1955 and Monthly Surface Water Sampling Reports for 1955 (second quarter) to 1964 (second quarter). After mid-1964, the Monthly Surface Water Sampling Reports were discontinued. However, a 1982 memo from Fee (the Y-12 Plant Manager) to Hickman (ORO) includes tables prepared by M. Sanders of mercury concentrations measured in weekly samples from 1954 to 1973 and monthly samples from 1973 to 1982 (Fee and Sanders 1982)**S** the 1983 Mercury Task Force estimates of mercury releases to EFPC were based on the data tabulated in this memo. For the years after 1982, mercury concentrations measured at Station E1 or Station 17 at the Y-12 site boundary, and presented in the ORR Site Environmental Reports, were used by the project team to calculate mercury releases to EFPC.

The Y-12 Plant Quarterly Reports and the Technical Division Monthly Progress Reports were two additional sources of quarterly and monthly averages of mercury concentrations in effluents to EFPC in the 1950s and 1960s. Individual values used to calculate these averages, or references for the individual

values, were not provided in these reports. However, data from these two report series were included in the project team's release calculation spreadsheets for comparison purposes (see Appendix I, Table I-3). For some periods, average concentrations presented in these reports did not agree with the averages calculated by the project team. This may be because averages reported in the Quarterly Reports and Monthly Progress Reports were often revised in the next quarter's report, apparently because of a lag in processing of the results for some of the samples, such that the average initially reported did not include all samples collected during that period.

For the period 1954 to 1964, the project team compared weekly sampling results gathered from memoranda and monthly sampling reports to the data tabulated by Fee and Sanders (1982) and used by the Mercury Task Force to estimate mercury releases to EFPC. The values listed under each month in the tables prepared by Sanders appear to be weekly composite measurements from memoranda for 1954-55 and from Monthly Surface Water Sampling Reports for 1955-64. The project team identified several apparent typographical errors made by Sanders in transcribing the values from the Monthly Surface Water Sampling Reports. In addition, for some months, the Sanders tables include additional data points in excess of the expected four or five weekly samples (Fee and Sanders 1982). For 1963-64, these extra data match values described in the surface water sampling reports as "EFPC downstream values". The project team did not use these data to calculate mercury releases to EFPC. For 1965-1982, however, the project team did not locate any other source of EFPC water concentrations and the weekly composite values could not be separated from any additional (*e.g.*, downstream) sample data that may have been collected. Consequently, the project team did not exclude additional values.

The project team was unable to locate Monthly Surface Water Sampling Reports for seven months during the period 1955 to 1964. In addition, weekly sampling results were not reported in the Monthly Surface Water Sampling Reports for approximately 15 weeks during this period. However, results for these weeks are reported by Fee and Sanders (1982). Consequently, the project team used the values reported in Fee and Sanders (1982) to fill in the gaps to calculate releases to EFPC.

Data used by the project team to calculate quarterly averages for 1955 to 1964 are presented in Appendix I, Tables I-2 and I-3.

Flow Rate Data

Prior to mid-1956, the project team located sporadic records of flow rate in EFPC. Beginning in mid-1956, EFPC flow was reported regularly in separate reports. The following sources of flow rate data were the primary sources used by the project team to calculate mercury releases to EFPC for the specified time periods:

- 1954 to 1955**S** In-plant Memoranda
- 1956 (3rd quarter) to 1964 (2nd quarter)**S** Monthly Surface Water Sampling Reports
- 1964 to 1971**S** Y-12 Quarterly Reports

- 1972 to 1985**S** Original recording charts and handwritten records of daily stream flow measurements
- 1986 to 1993**S** USGS daily measurements

The project team located flow data for the last quarter of 1954 and the second and third quarters of 1955, and used these flow rates to estimate actual pounds of mercury released to EFPC during these periods. For 1953, the first three quarters of 1954, and the first quarter of 1955, an average flow based on data available for 1954 through 1957 was used in the project team estimate. The project team used these data, along with mercury concentration data located for the second quarter of 1953 through the third quarter of 1955, to calculate pounds of mercury released to the creek instead of relying on inventory estimates.

Beginning in July of 1956, average weekly EFPC flow rates were included in Monthly Surface Water Sampling Reports. After surface water reports ceased around 1964, the project team used Y-12 Quarterly Reports as the source of EFPC flow rates**S** although the EFPC mercury concentrations were not reported in the Y-12 Quarterly Reports after 1962, flow rates were reported until 1971. For 1972-85, the project team used handwritten records of the original chart recordings of EFPC flow to calculate quarterly averages of EFPC flow rates. For the period 1986-93, the project team used daily USGS flow rate measurements extracted from a USGS database by the Y-12 HSEA Division Surface Water Group.

Calculated Releases and Comparison to 1983 Mercury Task Force Estimates

The project team's estimates of pounds of mercury released to EFPC are summarized below, and compared to the 1983 Mercury Task Force's estimates:

- For 1950-52, the project team's best estimate of mercury releases was 3,380 pounds, based on the mercury inventory estimate in the 1983 Mercury Task Force Report, since no mercury concentration data or flow rates were identified for this period. In their report, the 1983 Mercury Task Force assumed that 2.5% percent of the mercury inventory for each year was lost to water, based on an estimated average loss of Y-12 mercury inventory to water of 2.5% from 1955 to 1982. However, using concentration and flow rate data not identified by the 1983 Mercury Task Force, the project team calculated higher losses during 1953 and 1954, ranging from 2.9% to 7.3% of total inventory (Table 4-7). Based on these data, the percent of mercury inventory lost to EFPC during 1950 to 1952 was assumed to be between 3 and 8%.

Table 4-7: Percent of Mercury Inventory Estimated Lost to EFPC for 1950-1954

Year	Mercury Inventory (lb)	2.5% of Mercury Inventory (lb)	Conc/Flow Based Estimate (lb)	Conc/Flow Estimate as a % of Inventory	3%-8% of Mercury (lb)
1950	4,000	100	N/A	N/A	120-320
1951	8,000	200	N/A	N/A	240-640
1952	40,000	1,000	N/A	N/A	1,200-3,200
1953	162,000	4,050	11,799 [a]	7.3%	4,860-13,000
1954	241,000	6,025	7,057 [a]	2.9%	7,230-19,300

a Approximately 1,000 - 2,000 pounds of mercury were estimated to have been released during the first three quarters of 1953 and all four quarters during 1954. However, a very high mercury concentration reported for the 4th quarter of 1953 resulted in a release estimate of 9,000 pounds of mercury for that quarter. If the concentration for the 4th quarter 1953 was more typical of the other seven quarters, the percentage of the inventory released may have been closer to 3.2%.

- For 1953 and 1954, the project team's best estimate of mercury releases was 18,856 pounds, based on measured mercury concentrations and measured and estimated flow rates (an estimated average flow rate of 11 MGD for the 1950s was used for missing flow rates, based on the average flow rate for 1955 through 1957), compared to the 1983 Mercury Task Forces's estimate of 10,000 pounds based on the assumption that 2.5% of the Y-12 mercury inventory was lost.

- For the first three quarters of 1955, the project team's best estimate of mercury releases was 30,063 pounds, compared to zero releases reported for these three quarters in the 1983 Mercury Task Force Report. (The 1983 Mercury Task Force did not have stream flow data to calculate the loss quantity). During the fourth quarter of 1955, the best estimate of mercury releases was 5,793 pounds based on measured concentrations and flow rates.

- For 1956 to 1982, the project team's best estimate of mercury releases was 215,466 pounds, 2,478 pounds higher than the 1983 Mercury Task Force estimate due to identification of data missing when the Mercury Task Force Report was prepared and correction of math errors.

- For 1983 to 1993, the project team's best estimate of mercury releases was 468 pounds, based on water concentration and flow rate measurements.

In addition to the estimates of annual releases of mercury to EFPC, the project team included the metallic and storm correction factors developed in the 1983 Mercury Task Force Report (a total of 8,775 pounds) to the estimates of losses of mercury from the plant inventory. These correction factors were not included in the estimates of annual mercury releases to EFPC. The 1983 Mercury Task Force applied a correction of 7,500 pounds to their estimated total mercury release to EFPC to account for the reported observation that, in the 1950s, beads of mercury could be seen on the stream bottom. It was assumed that the method for sampling mercury in surface water did not record mercury that beaded and deposited to the sediment. The 1983 Mercury Task Force estimated that 15,000 pounds of mercury had accumulated in sediments since New Hope Pond was constructed in 1963, or an average of 750 pounds per year. The Mercury Task Force multiplied this number by 1.67 to account for the likely higher buildup of mercury between 1954 and 1959 than between 1963 and 1982. The resulting annual estimate of 1,250 pounds per year was then multiplied by 6 years (1954-59) to produce a total "metallic mercury" correction factor of 7,500 pounds.

The 1983 Mercury Task Force applied an additional correction of 1,275 pounds to the estimated mercury release to EFPC to account for collection of samples from 1963-82 on a time-proportional rather than a flow-proportional basis. Weekly grab samples between 1977 and 1982 indicated that the mercury release rate was approximately 13% greater during rain events. Therefore, the 1983 Mercury Task Force multiplied the total quantity of mercury measured in EFPC samples between 1963 and 1982 (9,851 lb) by a "storm correction factor" of 13% to incorporate the additional mercury released during higher creek flows. This correction factor was used in the project team's estimate of losses from the plant inventory of mercury, but not in estimates of mercury released to water.

The project team's best estimate of the total mass of mercury released to EFPC between 1950 and 1993 (to two significant figures) is 280,000 pounds. By comparison, the 1983 Mercury Task Force estimate was 240,000 pounds. This is an increase of 40,000 pounds or 17%. Table 4-8 summarizes differences between the estimates by the Task 2 project team and the 1983 Mercury Task Force.

Table 4-8: Comparison of Mercury Task Force and Dose Reconstruction Team Estimates of Mercury Released to EFPC

Year	1983 Task Force Estimate (lb)	1996 Task 2 Estimate (lb)	Difference (lb)	Difference (%)
1950	100	260	+160	160
1951	200	520	+320	160
1952	1,000	2,600	+1,600	160
1953	4,000	11,799	+7,799	190
1954	6,000	7,057	+1,057	18
1955	5,881	35,856	+29,975	510
1956-1982	212,988	215,466	+2,478	1
1983-1993	Not applicable	468	+468	Not applicable
Corrections for metallic Hg and effects of storms	+8,775	+8,775	0	0
TOTALS (To 2 significant figures)	**238,944 (240,000)**	**282,801 (280,000)**	**+43,857 (+40,000)**	**+17**

Figure 4-4 graphically presents the comparison of the 1983 and 1996 estimates of mercury released to EFPC.

4.5.2 Uncertainty in Measurements of Concentration and Water Flow Rate

Uncertainty in the measurements of mercury concentrations in EFPC was assumed to range from ± 10% to ± 50% of the reported concentration, depending on the method used and the nearness of the measured concentration to the method limit of detection achievable during a particular year. These uncertainty estimates were taken directly from references for the analytical methods discussed in Appendix B, and are described in Table I-1 (Appendix I).

Uncertainty in water flow rates determined at the weir in EFPC near the Y-12 warehouse prior to 1963 is estimated to be ± 15%. After 1963, the uncertainty in flow rates measured at the weir located at the outfall of New Hope Pond is estimated to be ± 10%, based on ratings of the quality of flow measurements at the ORR provided by USGS[1].

[1]Private communication between Brian Caldwell of the project team and Bradley Bryan of the U.S. Geological Survey. July 1996.

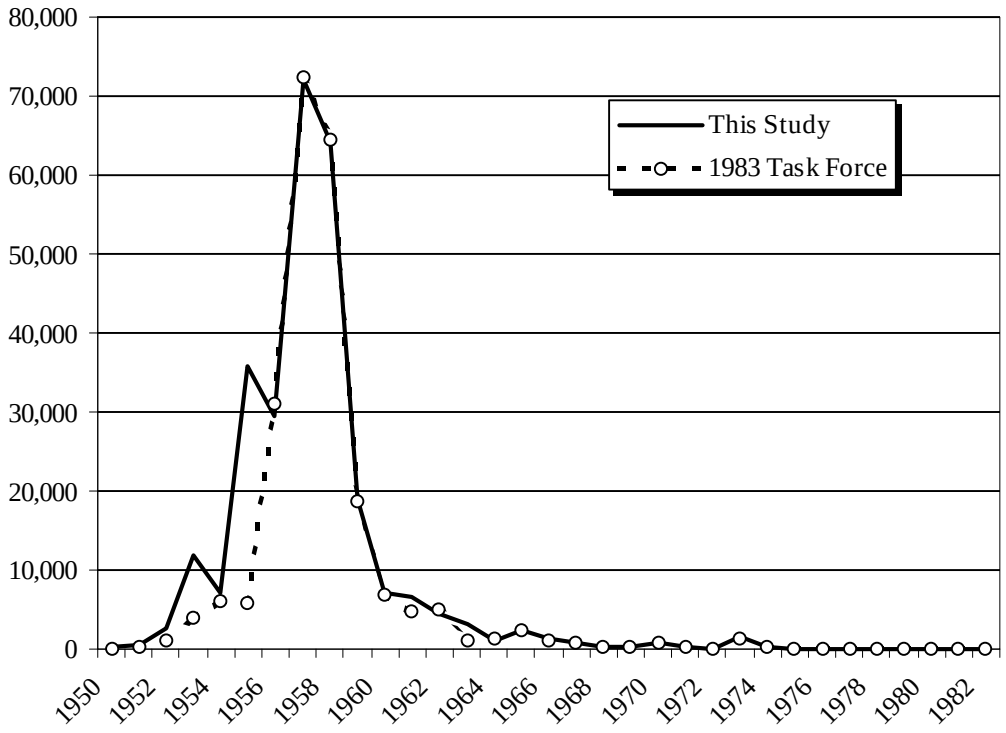


Figure 4-4: Annual Waterborne Mercury Release Estimates by the Task 2 Project Team and the 1983 Mercury Task Force (pounds)

4.5.3 Documentation of Calculations and Numerical Results

Appendix I documents the calculations performed by the project team in the tables and spreadsheets described below:

- Table I-1 summarizes concentration and flow rate data used to estimate releases of mercury to EFPC for each year from 1950-90, and the uncertainty associated with the measurement of the concentrations and flow rates.

- Table I-2 presents the three sources of EFPC mercury concentration and flow rate data for 1950-90, and the values chosen for Task 2 estimation of mercury releases to EFPC. When available, the lowest level of reported summary data (i.e., weekly composite rather than monthly or quarterly average) was used. This table also shows the calculations used to check the 1983 Mercury Task Force Report calculations of mercury released, and a comparison of the 1983 estimates of mercury released to EFPC with the project team's estimates.

- Table I-3 shows all mercury concentration, flow rate, and pH data available, and documents the project team's calculation of monthly and quarterly averages.

- Table I-4 presents a comparison between data obtained from Fee and Sanders (1982) that were used to estimate mercury releases to water in the 1983 Mercury Task Force Report, and data used by the project team to estimate releases. Table I-5 contains the calculations for the data presented in Fee and Sanders (1982).

4.5.4 Mercury Speciation in Y-12 Liquid Effluent

The 1983 Mercury Task Force Report is the only document located by the project team that refers to the chemical and physical forms of mercury released from Y-12 into EFPC or into the ambient air. The 1983 Mercury Task Force Report information is not based on chemical and physical analysis, but on process knowledge and professional judgement. Detailed discussions of the forms of mercury assumed to have been released are presented in Appendix B.

Losses to water (i.e., EFPC) are largely traceable to a process waste stream resulting from acid washing of mercury. The operation responsible for generating this waste was essential to the operation of the lithium separation process, but was modified in June 1958 to reduce mercury losses and continued through 1961. In the period before 1961, about 200,000 pounds of mercury were discharged to the creek from the Colex process as a very dilute (ppm of mercury) neutralized nitric acid process waste stream (not as elemental mercury). It was the opinion of the 1983 Mercury Task Force that the initial form of the mercury was soluble or a very finely divided suspension of mercuric oxide. The 1983 Mercury Task Force based this opinion on the fact that mercuric nitrate, which would have been produced when the mercury was washed with nitric acid, is very soluble in water. However, neutralization of the waste stream would have formed mercuric oxide, which is only slightly soluble. Mercuric oxide formed in this manner in the relatively dilute concentrations involved here does not settle readily, and flowing water would likely have kept it in suspension. Further, mercury in surface water has a strong affinity for sediment and particulate matter, and a significant fraction of the mercury released would likely have sorbed on finely divided particulate matter, both organic and inorganic, and been either deposited to sediment or transported further downstream (Horowitz 1991).

According to analytical information in the 1983 Mercury Task Force Report (UCCND 1983a), EFPC water was historically analyzed for total mercury (by the Y-12 Plant laboratory) except for a few years in the mid-1970s. During the period from January 1974 to June 1977, the water samples from EFPC were analyzed for soluble mercury only. The insoluble loss for this time period was estimated and included in the 1983 Mercury Task Force estimate. Mercury (soluble) concentrations in the filtrate from New Hope Pond samples were less than the detectable limit, indicating that mercury was being discharged predominately in suspended (insoluble) form.

4.6 Mercury Released Through Spills to the Ground at Y-12

Regarding the uncertainty of estimates of mercury lost from large spills and the contribution, if any, of mercury from spills to concentrations in EFPC, the 1983 Mercury Task Force Report states that for four of five of the large spills they recovered mercury that was on a floor or solid surface, but could not recover most of what went into the dirt below building floors. Some dirt with visible mercury was dug up and sent to the 81-10 mercury recovery facility, but the source of the dirt was not tracked and there was no quantification of how much mercury was eventually recovered at 81-10 from each spill. It should be noted that over 3.5 million pounds of mercury were recovered as a result of operations at Building 81-10. The 425,000 pounds of mercury lost to the ground through spills was probably all metallic mercury (UCCND 1983a).

Loss estimates for these four spills were apparently based on classified flow rates and time durations of the events (e.g., the wrong valve being open). For the first four spills, there are no reports or documentation of the flow rates or leak times to evaluate. The fifth large spill was from a tank, with a sight gauge used to estimate the loss by difference. The best estimate of loss is 49,853 pounds, with a lower bound (tanks could have been almost empty just before the spill) of 44,853 pounds, and an upper bound (tanks could have been full) of 153,245 pounds. The loss estimates from all five of the spills are upper bound estimates, because the mercury recovered from them was not quantified. Any mercury from the spills that reached EFPC would be included in the discharge measurements, or trapped in sediments at the bottom of the watercourse upstream from the discharge monitoring weir.

5.0 MEASUREMENTS OF MERCURY IN THE ENVIRONMENT NEAR THE ORR

Numerous investigators have conducted studies of mercury in the environment near the ORR. This section describes:

- Studies that characterized mercury *concentrations* in the environment near the ORR, and
- Studies that characterized the *forms*, or *species*, of mercury in the environment near the ORR.

In subsequent sections, information from these studies is combined with historical release data to estimate exposures to off-site populations.

5.1 Historical Environmental Monitoring Programs for Mercury

Exposures to mercury released from the ORR were likely most significant between 1950 and 1963, when lithium was being processed at Y-12 and airborne and waterborne releases were highest. While the Task 2 team identified approximately 50 historical studies describing concentrations of mercury in the environment near the ORR, most of these studies were conducted after 1970. Locations where environmental monitoring for mercury was conducted before 1970 include surface water monitoring at the EFPC/Poplar Creek junction, Poplar Creek, and the Clinch River from 1955 to 1962, and at the Y-12 release point on EFPC beginning in 1953.

Studies describing concentrations of mercury in air, surface water, soil, sediment, plants, animals, and fish and other aquatic biota near the ORR, relevant to evaluating mercury releases from Y-12, are summarized in Tables 5-1 through 5-7. Environmental monitoring programs for mercury near the ORR of particular significance are described in greater detail in Appendix J, as are concentrations of mercury measured in these studies. Examples of background concentrations of mercury in various environmental media are presented in Table 5-8 for comparison purposes. Most of the concentrations presented do not reflect pristine conditions, since redistribution of mercury air emissions has resulted in global mercury contamination.

5.2 Speciation of Mercury in Environmental Media near the ORR

Mercury exists in the environment in a number of different chemical forms, or species, affected by the physical, chemical, and biological conditions of the environment. Although lithium enrichment processes at Y-12 used elemental mercury (Hg^0), chemical and physical processes in the environment have caused conversion of the mercury to other forms. Understanding the species of mercury in the environment is particularly important because the way mercury moves through the environment, the likelihood it will

Table 5-1: Historical Air Monitoring Programs for Mercury Near Y-12

Year(s)	Description of Samples	Location(s) of Sample Collection	Reference
1953-1983	Building air monitoring	Y-12	UCCND (1983a)
1986-1989	Ambient air monitoring	Y-12 (east end)	MMES (1991a)
1987-1989	Ambient air monitoring	Near Lake Reality	MMES (1991a)
1988-1989	Ambient air monitoring	• Chestnut Ridge • Walker Branch Watershed	MMES (1991a)
1991-1992	Ambient air monitoring	Three EFPC floodplain sites and a control site atop Chestnut Ridge	Turner and Bogle (1992)

Table 5-2: Historical Surface Water Sampling Programs for Mercury Downstream of Y-12

Year(s)	Description of Samples	Location(s) of Sample Collection [a]	Reference
1953-1983	Weekly composite of direct releases to EFPC	Y-12 release point	UCCND (1983a)
1955-1961	Weekly sample from each location	• Confluence of EFPC/ Poplar Creek • Poplar Creek • Clinch River	K-25 Technical Division (Kwasnoski and Whitson 1955-1961)
1970	Single sample from each location	• New Hope Pond • EFPC (Mile 6.5, 9.7, 14)	Sanders (1970)
1971–present	Routine reservation-wide environmental monitoring program (monthly composite from each location)	• Clinch River below Poplar Creek (variable locations) • EFPC below New Hope Pond (outflow)	UCC (1972-1982) MMES (1984-1991a)
1985	Single sample from each location	• EFPC (Mile 14.36) • Poplar Creek (Mile 13.8) • Clinch River (Mile 6.8, 24.0)	TVA Instream Contaminant Study (TVA 1985a)
1989-1990	Single sample from each location	• Poplar Creek (Mile 0 to 5.5) • Clinch River (Mile 0 to 12) • Watts Bar Reservoir	Clinch River Remedial Investigation (Cook et al. 1992)
1991-1992	One to two samples from each location	EFPC (Mile 0, 3, 7, 12, 14)	EFPC-Sewer Line Beltway Remedial Investigation (SAIC 1994b)

a River miles are measured from the mouth of a river or stream upstream to its source

Table 5-3: Historical Soil Sampling Programs for Mercury Near Y-12

Year(s)	Description of Samples [a]	Location(s) of Sample Collection	Reference [b]
1983-1987	Surface soil	• EFPC floodplain • Oak Ridge community	ORAU environmental monitoring and surveillance studies (TDHE,1983; Hibbitts 1984; Hibbitts 1986)
1985	Soil core (depth approx. 3 feet)	• Poplar Creek bank near Blair Road bridge	Olsen and Cutshall (1985)
1990-1992	• Phase Ia, soil cores from NOAA, Bruner's Center, and Sturm sites • Phase Ib, soil cores from 159 transects across EFPC floodplain at 100 m intervals, composited for depth intervals from 0-16 in., 16-32 in, and 32-48 in. bgs	• EFPC floodplain • Sewer Line Beltway	EFPC Floodplain/ Sewer Line Beltway RI/FS (SAIC 1994b)

a bgs = below ground surface

b ORAU = Oak Ridge Associated Universities; RI/FS = remedial investigation/feasibility study

Table 5-4: Historical Sediment Sampling Programs for Mercury Downstream of Y-12

Year(s)	Description of Samples	Location(s) of Sample Collection	Reference
1970	10 mud samples	• New Hope Pond • EFPC (exact locations not given)	Sanders (1970)
1970s	Single surface samples	• EFPC • Poplar Creek • Clinch River	Reece (1974)
1970s	Single surface samples	• Poplar Creek • Clinch River	UCCND (1983a)
1974-1976	Single surface samples	• Poplar Creek • Clinch River	Elwood (1977, 1984)
1975-1981	• Surface samples (2 times yr^{-1})	• EFPC (Mile 0.5) • Poplar Creek [Mile 0 to 5, (17 locations)]	ORGDP (1981)
1982	• Core from New Hope Pond to evaluate Hg deposition since last major dredging activity in 1973 • Single surface samples in EFPC	• New Hope Pond (core) • EFPC [Mile 1.3 to 14.2 (8 locations)]	Van Winkle et al. (1984)
1985	Shallow cores	Watts Bar Reservoir and other E. Tenn. reservoirs	TVA (1986)
1984	Cores to evaluate transport of Hg in sediment during stormflow events	• EFPC floodplain (122 locations) • EFPC (19 locations) • Poplar Creek (3 locations) • Clinch River (8 locations) • Watts Bar Reservoir (7 locations)	TVA (1985b, c,d)
1985	• 180 surface samples • 3 cores	• EFPC • Poplar Creek • Clinch River	Ashwood et al. (1986)
1985	Two 1-meter cores	Poplar Creek at proposed construction site of Blair Road Bridge (near K-25 Plant) (one in floodplain, one in creek bed)	Olsen and Cutshall (1985)
1986	• 190 surface samples • 60 cores	Watts Bar Reservoir	Olsen et al. (1990)
1989-1990	• Surface grab samples • Cores	• Poplar Creek • Clinch River • Watts Bar Reservoir	CRRI (Cook et al., 1992)
1990	• 5 composited 12-inch cores • Cores from Watts Bar Reservoir water intakes	12 recreational areas on Watts Bar Reservoir	TVA (1991b)

Table 5-5: Historical Vegetation Sampling Programs for Mercury Near Y-12

Year(s)	Description of Samples	Location(s) of Sample Collection	Reference
1982	Live/ dead pasture grass at distances of 5, 30, and 100 m from the stream bank	EFPC (Miles 5.5 and 8.3)	Van Winkle et al. (1984)
1983-1987	100 co-located soil and plant sample pairs	Throughout the city of Oak Ridge and EFPC floodplain	TDHE (1983), Hibbitts (1984), Hibbitts (1986)
1992	16 co-located soil and plant sample pairs	Bruner's Center in EFPC floodplain	SAIC (1994b)

Table 5-6: Historical Terrestrial Biota Sampling Programs for Mercury Near Y-12

Year(s)	Description of Samples	Location(s) of Sample Collection	Reference
1982	Tissue from cow	Grazed in EFPC floodplain	UCCND (1983a)
1982	Hair from cow and horse	Grazed in EFPC floodplain and drank water from EFPC	UCCND (1983a)
mid-1980s	Muscle and liver tissue from deer killed in vehicle collisions	Oak Ridge Turnpike adjacent to EFPC	Gist (1987) as cited in Travis et al. (1989)

Table 5-7: Historical Fish Sampling Programs for Mercury Downstream of Y-12

Year(s)	Description of Samples	Location(s) of Sample Collection	Reference
1970	• Carp and bluegill	• Pond near EFPC • EFPC (Mile 14.2)	Sanders (1970)
1976	• 6 species • 272 samples	• Poplar Creek (Mile 0 to 6.0) • Clinch River (Mile 4.5 to 13.5)	Elwood (1984)
1977, 1979	• 15 species	• Poplar Creek (Mile 0.5, 5.5, 11.0) • Clinch River (Mile 10.5, 11.5, 15, 19, 21.9)	Loar et al. (1981a,b)
1982	• 14 species	• Poplar Creek	Stiff (1982)
1982	• 3 species	• EFPC (Mile 1.3, 8.3, 14.1, 14.2)	Van Winkle et al. (1984)
1983	• 6 species	• Ponds near EFPC	Blaylock et al. (1983)
1984	• 11 species	• Poplar Creek (Mile 0.2) • Clinch River (Mile 2, 6, 11, 20) • Tennessee River (Mile 572, 558)	TVA (1985e)
1990	• 3 species	• Poplar Creek (Mile 1.4, 4.6, 5.3) • Tennessee River (Mile 530.5, 557.0)	Cook et al. (1992)

Table 5-8: Background Environmental Concentrations of Mercury

Medium (units)	Concentration	Year	Comments
Ambient Air (µg m^{-3})	0.002 - 0.010	1984	average concentration (USEPA Mercury Health Effects Update)
	0.0055	1988-89	over the forested Walker Branch, TN (mean concentration)
	40 - 80	1971	stack gas of large coal-fired power plants in Ontario, Canada
Surface Water (µg L^{-1})	0.001 - 0.003 [1]	1986	unpolluted inland lakes in Canada
	< 0.005	1991	freshwater with no known sources of mercury contamination
	0.008 - 0.017	1991-92	rainwater collected in Walker Branch, Tennessee watershed
	0.0005 - 0.104	1990	California lakes and rivers
	0.005 - 0.100	1984	drinking water (USEPA Mercury Health Effects Update)
Soil (mg kg^{-1})	0.08	1970	approximate concentration in the earth's crust
	0.020 - 0.150	1972	soil and glacial deposits in Canada
	0.040 - 0.193	1983-84	coastal North Carolina
	0.010 - 0.550	1994	New York State orchard soils
	0.020 - 0.625	1979	virgin and cultivated soils from a number of countries
	<0.01 - 3.4	1986	Eastern USA soils
Sediment (mg kg^{-1})	0.008 - 0.020	1983-84	Pungo River, North Carolina
	0.090 - 0.210 [1]	1980-84	5 lakes and river sites in Finland
	0.090 - 0.240	1989	Wisconsin lakes due to atmospheric deposition
Fish (mg kg^{-1}, wet wt.)	0.153 [1]	1972	freshwater fish from US Fish and Wildlife National Pesticide Monitoring Program (mean concentration)
	0.112 [1]	1976	freshwater fish from US Fish and Wildlife National Pesticide Monitoring Program (mean concentration)
	0.110 [1]	1980, 1984	freshwater fish from US Fish and Wildlife National Pesticide Monitoring Program (mean concentration)
	0.17 - 1.8 [1]	1986-89	predatory game fish from USEPA study of residues in fish
	0.05 - 0.200 [1]	1985	most freshwater fishes; <0.010 in short-lived herbivorous species
	0.290 - 1.69 [1]	1980-84	5 lakes and river sites in Finland
	3.41 [1]	1992	bluefin tuna in the Northwest Atlantic Ocean (mean conc.)
Plants (mg kg^{-1}, dry wt.)	0.00001 - 0.037	1972	terrestrial plants in Canada
	0.200 - 30.0	1972	terrestrial plants in Canada near natural mercury deposits

Notes: [1] Mercury as methyl or organic mercury. All other values are total mercury.

Sources: Toxicological Profile for Mercury- Draft for Public Comment (August 1997), U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry (ATSDR).
Files for Mercury (CAS# 7439-97-6) and Methylmercury (CAS# 22967-92-6), Hazardous Substances Data Bank (HSDB) 1998, National Library of Medicine, Washington, D.C.
Elements in North American Soils (1991). J. Dragun and A. Chiasson. Hazardous Materials Control Resources Institute, Greenbelt, MD.

be taken into the body on contact, and how it behaves once in the body are dependent on the forms of mercury that are present in the environment and that are contacted.

When evaluating health risks from environmental exposures, information on chemical speciation can be used to:

- Predict how the chemical will behave in the environment,
- Identify appropriate reference criteria to describe the likelihood that the chemical will produce an adverse health effect, and
- Predict how much of the chemical will be absorbed into the body following an environmental exposure relative to how much was absorbed under the exposure conditions that are the basis for the health effects reference criteria.

Mercury species found in the environment or used in industrial processes are usually classified into three groups. These groups are described below.

Metallic or elemental mercury (Hg^0), a shiny, silver-white, extremely dense, odorless liquid, is the familiar mercury species found in thermometers. Some evaporation of elemental mercury occurs at room temperature to form mercury vapor. Compared to some mercury compounds (e.g., mercuric chloride, mercuric acetate), elemental mercury is relatively insoluble in water.

Inorganic mercury compounds or "salts" [including mercuric sulfide (HgS), mercuric chloride ($HgCl_2$), mercuric hydroxide ($Hg(OH)_2$)] form when mercury ions (such as divalent mercury, Hg^{2+}) combine with other elements such as chlorine or sulfur or with hydroxide (OH^-) ions in aqueous solution. Elemental mercury and inorganic mercury compounds are often grouped under the generic term "inorganic mercury" (in this report, "inorganic mercury" refers to mercuric mercury salts, while "elemental mercury" refers to Hg^0). The water solubility of inorganic mercury compounds ranges from nearly insoluble (HgS) to highly soluble ($HgCl_2$).

Organic mercury compounds, including methylmercury (CH_3Hg^+) and dimethylmercury (CH_3CH_3Hg), form when mercury combines in a chemical bond with carbon. Bacteria and abiotic (chemical) processes can methylate mercury(II) ions (that is, add a methyl (CH_3) group) to form methylmercury compounds. Methylmercury is more easily absorbed by fish and other aquatic fauna than elemental and inorganic mercury, and can bioaccumulate to higher concentrations than in surrounding media.

Typically it is assumed that, once taken into the body and absorbed into the blood, compounds *within* each of these three groups have the same critical health effect endpoint (the first adverse effect that occurs with increasing dose).

Differences in chemical and physical characteristics of mercury compounds within each group, however, will have an effect on how much of the mercury is in a form that can be absorbed into the bloodstream. For example, the solubility of inorganic mercury compounds varies widely (Table 5-9), and highly soluble

forms (such as mercuric chloride) are more readily absorbed into the bloodstream following ingestion or other contact than insoluble forms (such as mercuric sulfide). In other words, the soluble forms are more *bioavailable.*

Table 5-9: Water Solubility of Mercury Species and Compounds (at about 20 C)

Compound	Solubility in Water
Mercuric acetate	400,000 mg L^{-1} (a)
Mercuric chloride ($HgCl_2$)	70,000 mg L^{-1} (b)
Mercuric oxide (HgO)	53 mg L^{-1} (b)
Elemental mercury (Hg^0)	0.060 **S** 0.080 mg L^{-1} (b)
Mercuric sulfide (HgS)	~0.010 mg L^{-1} (b)

a ATSDR (1994)
b Henke et al. (1993)

Quantifying the species of mercury in the environment at any given time can be difficult because mercury converts to other forms under different environmental conditions and because extraction and analytical methods used to measure the quantity of mercury in an environmental sample may change the speciation (Davis et al. 1996). The species of mercury can, however, be predicted from information about the species of mercury that is released and the characteristics of the environmental medium. For example, elemental mercury (Hg^0) used in the Colex process at Y-12 was washed with nitric acid, likely causing its conversion to divalent mercury (Hg^{2+}) through a process called oxidation in which electrons are lost from the elemental mercury and the mercury becomes positively charged (Hg^0 ÷ Hg^{2+}). Chlorine in chlorinated process and cooling water may also have caused the oxidization of elemental mercury. Once oxidized to Hg^{2+}, sulfate-reducing bacteria, such as found in sediments, can cause the Hg^{2+} to form bonds with other elements or chemical groups, such as sulfur (S^{2-}) to form mercuric sulfide (HgS) or a methyl group (CH_3^-) to form methylmercury (CH_3Hg^+).

Typically, toxicity benchmark values used for regulatory purposes to predict the potential for adverse health effects, including reference doses (RfDs) established by the U.S. Environmental Protection Agency (USEPA) or minimal risk levels (MRLs) established by the Agency for Toxic Substances and Disease Registry (ATSDR), are based on studies with laboratory animals or on studies that investigate worker exposures (known as epidemiologic studies). The species of chemical investigated in these studies is not always the same as the chemical species associated with environmental exposures.

The species of mercury used as the bases for toxicity benchmarks for different routes of exposure to mercury, and examples of these values, are summarized in Table 5-10. These toxicity benchmarks are described in greater detail in Section 11.0.

Table 5-10: Species of Mercury Commonly Evaluated in Investigations of Toxicity through Different Routes of Exposure

Exposure Route	Mercury Species Commonly Investigated in Toxicity Studies	Example(s) of Toxicity Benchmark Values
Inhalation	Elemental mercury (Hg^0) vapor	• USEPA Reference Concentration (RfC), based on long-term exposure of workers to airborne mercury vapors (IRIS 1998) • ATSDR Minimal Risk Level (MRL), based on acute exposures of young rats (11-17 days old) to airborne mercury vapors (ATSDR 1997)
Ingestion	Soluble inorganic mercury (Hg^{2+}) salts	• USEPA Reference Dose (RfD) and ATSDR MRL, based on intermediate-duration exposures of rodents to mercuric chloride administered in food or water (IRIS 1998; ATSDR 1997)
Ingestion	Methylmercury	• USEPA RfD, based on ingestion of methylmercury-treated grain by pregnant women in Iraq, and effects on children exposed *in utero* (IRIS 1998) • ATSDR MRL, based on ingestion of methylmercury in fish by pregnant women in the Seychelles Islands, and effects on children exposed *in utero* (ATSDR 1997) • Ongoing studies are investigating the health effects of ingestion of methylmercury in fish (Crump et al. 1996; Grandjean et al. 1992; Kjellstrom et al. 1989; McKeown-Eyssen et al. 1983; Marsh et al. 1995; Myers et al. 1995)

Often, toxicity benchmarks are based on studies of health effects following administration of much higher doses than typically occur in environmental exposures. For example, much of the toxicological data on adverse health effects associated with exposures to inorganic and elemental mercury are based on studies of animals administered high doses in the laboratory or on data from humans exposed to high mercury concentrations in occupational settings. Likewise, a significant portion of the database describing toxic effects from exposures to methylmercury is based on exposures to methylmercury not methylated in the environment (specifically exposures of a population in Iraq who consumed seed grain that had been treated with methylmercury because of its fungicide properties). Because these are the most robust data sets available, it is necessary to compare typically lower dose environmental exposures to dose-response relationships established using these data.

Since the routes of exposure and chemical species investigated in environmental assessments often differ from the toxicity studies to which they are compared, the internal (absorbed) doses generally differ even when the external (administered) doses are the same. These differences in the amount of a chemical that is bioavailable through different routes of exposure are accounted for in the dose equation, described in Section 2.1, by the *relative bioavailability* parameter. The relative bioavailability parameter can be described as the ratio between the bioavailability of a substance in a person exposed in the environment and the bioavailability of that substance administered to an animal in a toxicity study or to which a human is exposed in a worker exposure study. Although knowledge of mercury speciation in air, soil, water and other environmental media at Oak Ridge and other sites is imperfect, due to a number of factors including analytical difficulties and temporal and spatial variability, speciation information is sufficient to justify selection of a toxicity benchmark value from among those described in Table 5-10, to which to compare estimates of mercury exposures due to releases from the ORR.

For example, data on mercury speciation in soil in the EFPC floodplain indicate it is a mixture of inorganic and elemental mercury species; therefore, it is most appropriate to compare exposures from ingestion of soil to toxicity data based on studies of ingestion of inorganic mercury compounds (see Table 5-10). Very little data are available addressing the toxicity of elemental mercury following ingestion, in part because elemental mercury tends to be poorly absorbed following ingestion and because ingestion is not a common route of exposure to elemental mercury. In this assessment, uncertainties about the relative bioavailability of mercury species in an environmental medium compared to the bioavailability in the reference toxicity study are reflected in the PDFs used to characterize the relative bioavailability parameter. PDFs were selected to account for both uncertainties in speciation data as well as expected temporal and spatial variability in speciation within a medium.

The sections that follow summarize:

- Studies by investigators at Oak Ridge and elsewhere to qualitatively or quantitatively characterize the species of mercury in different environmental media,
- Mercury species assumed by the project team to be present in each environmental medium, for purposes of comparing estimated doses to toxicity benchmarks, and
- Relative bioavailability PDFs established by the project team for each medium and route of exposure.

5.2.1 Mercury Speciation in Surface Water

As described in Section 3.5, mercury was released from the Y-12 Plant directly to surface water (EFPC). The 1983 Mercury Task Force Report (UCCND 1983a) is the only document located by the Task 2 project team that refers to the chemical and physical forms of mercury released from Y-12 into EFPC (see Appendix B).

Data Describing Mercury Speciation in Surface Water Downstream from Y-12

Data describing the speciation of mercury in surface water downstream from the ORR and at other sites indicate:

- In the period before 1961, about 200,000 pounds of mercury were discharged to EFPC from a process in which elemental mercury used in the lithium separation process was cleaned with a nitric acid solution. The goal of this process was to remove impurities detrimental to the separation process. Treatment with nitric acid likely oxidized some of the elemental mercury to the mercuric state (Hg^{2+}), greatly increasing both the solubility and chemical reactivity of the discharged mercury, but decreasing its volatility. In the oxidized state, mercury has a strong tendency to (1) bind rapidly to suspended matter, (2) combine with the sulfide ion and sulfur-containing compounds to form highly insoluble compounds, or (3) be reduced again to the elemental state[1].

- During the 1950s and 1960s, there was abundant suspended matter available in EFPC, including organic matter and coal fines, to bind oxidized mercuric mercury (Hg^{2+}). In addition, regular back flushing of sand filters at the Oak Ridge Water Treatment Plant located on Pine Ridge near the east end of Y-12 supplied suspended matter from the Clinch River (the source of water to the treatment plant) to the upper reach of EFPC (Wing unpublished)[1].

- During the 1950s, sulfide levels in EFPC were probably high. Sulfide sources likely included organic loading, which fosters microbial reduction of sulfate to produce elevated sulfide levels in sediments, and release of sulfides from acidification of metal sulfide compounds, such as may have occurred when the pH of EFPC varied widely prior to construction of New Hope Pond in 1963. Sulfide is a powerful precipitant for mercuric mercury, leading to production of insoluble mercuric sulfide.[1]

[1] Personal communication between T. R. Mongan and G. M. Bruce of the project team with Ralph Turner and Nicolas Bloom of Frontier Geosciences, Inc., May 1997.

- Although elemental mercury is somewhat soluble in water (about 60 parts per billion (ppb), equivalent to 0.060 mg L^{-1}), it could not have accounted for the concentrations of total mercury observed in EFPC in the 1950s. Annual average mercury concentrations measured in EFPC (up to 3 parts per million (ppm), equivalent to 3 mg L^{-1}) far exceeded the solubility of elemental mercury. At the maximum concentrations measured in EFPC, the solubility of elemental mercury would have limited it to less than 2% of total mercury concentrations.

Mercury speciation significantly impacts the fate of mercury in surface water systems— whether the mercury evades (volatilizes) from the water surface, binds to suspended particulates or sediment, or remains dissolved in water. Mercury in any surface water system likely consists of a mixture of dissolved inorganic mercury, dissolved elemental mercury, mercury sorbed to particulates, and a very minor fraction of dissolved organic mercury. A number of water quality parameters, including pH and levels of suspended solids, as well as the volume of available mercury, can affect the equilibrium between these forms.

For evasion of mercury from surface water to occur, the mercury must exist in a volatile form such as elemental mercury or dimethylmercury. Studies in lakes and oceans have shown that dissolved elemental mercury (referred to as dissolved gaseous mercury or DGM) accounts for less than 1% to about 6% of the total dissolved or suspended mercury in these systems (Liebert et al. 1991, Amyotl et al. 1995, Saouter et al. 1995).

DGM may form through several processes including:

- Solubilization of elemental mercury in contact with the water (the solubility of elemental mercury in water is about 60 ppb),

- Reduction of divalent mercury (Hg^{2+}) to elemental mercury by both biotic and abiotic processes ($Hg^{2+} \div Hg^{0}$), and

- Dismutation of monovalent mercury to divalent mercury and elemental mercury ($2Hg^{+} \div Hg^{2+} + Hg^{0}$).

While the relative importance of individual processes can vary considerably from site to site and from time to time, these processes collectively regulate the fraction of mercury available to escape from water by evasion.

The project team identified no data on measurements of dimethylmercury or other organic mercury species in water downstream of Y-12. Studies of mercury speciation in surface water at other sites with high mercury concentrations, including the Carson River in Nevada, show that organic mercury compounds typically comprise less than 5% of mercury in surface water (Praskins 1996; Henke et al. 1993; Porcella 1994).

Based on this information, the project team assumed that mercury in surface water near the ORR was a mixture of dissolved inorganic and gaseous (elemental) mercury and suspended inorganic mercury sorbed to particulates.

Selection of Toxicity Benchmarks and Relative Bioavailability Factors for Evaluating Exposures to Mercury in Surface Water

As indicated in Table 5-10 and described in detail in Section 11.0, most studies of inorganic mercury toxicity following ingestion evaluate highly soluble inorganic mercury salts dissolved in water. Doses from ingestion of mercury in EFPC water were therefore compared to toxicity benchmarks based on ingestion of inorganic mercury. Doses were evaluated assuming that the bioavailability of mercury in EFPC water may have ranged from slightly less than the bioavailability of the mercury in the toxicity studies (because elemental mercury and mercury sorbed to particulate are less soluble than the inorganic forms of mercury evaluated in the toxicity studies) to as bioavailable as the mercury in the toxicity studies. The relative bioavailability PDF for evaluating ingestion of mercury in water ($B_{oral\text{-}water}$) was characterized by a uniform distribution with a minimum of 0.8 and a maximum of 1.0.

5.2.2 Mercury Speciation in Air

In addition to direct releases to surface water, mercury was released from Y-12 directly to air. Mercury may also have volatilized from contaminated surface water and soil.

Data Describing Mercury Speciation in Air Near Y-12

Data describing the speciation of mercury in air near the ORR and at other sites indicate:

- Mercury was released to air from ORR operations largely as elemental mercury vapor (UCCND 1983a).

- Greater than 99% of airborne mercury in the Walker Branch watershed of the Clinch River (located just southwest of the Y-12 Plant on the south side of Chestnut Ridge) occurs as vapor (Lindberg et al. 1991), with less than 1% occurring as particulate. Similar percentages have been reported for other sites (Fitzgerald and Clarkson 1991; Mason et al. 1994).

- Other forms of mercury (including inorganic mercury and methylmercury) typically comprise less than 2% of the total airborne concentration (Fitzgerald 1986; Fitzgerald 1989; Fitzgerald and Clarkson 1991; Henke et al. 1993).

Based on this information, the project team assumed that airborne mercury near the ORR was elemental mercury vapor.

Selection of Toxicity Benchmarks and Relative Bioavailability Factors for Evaluating Exposures to Mercury in Air

As indicated in Table 5-10 and described in detail in Section 11.0, most studies of the inhalation toxicity of mercury are based on worker exposures to mercury vapor. Doses from inhalation were therefore compared to toxicity benchmarks based on inhalation of airborne mercury. Because the chemical and physical characteristics of mercury in breathing zone air near the ORR are assumed to be similar to those in these worker studies, doses from inhalation of airborne mercury were evaluated assuming that the bioavailability of mercury in air near Oak Ridge was the same as in the worker studies (i.e., the *relative* bioavailability is 1.0). The relative bioavailability PDF for evaluating inhalation of mercury in air (B_{inh}) was characterized by a point estimate with a value of 1.0.

5.2.3 Mercury Speciation in Soil/ Sediment

The speciation of mercury in soil is very complicated. Soils and sediments in the EFPC floodplain likely contain a mixture of elemental and inorganic mercury; however, analytical methods can strongly affect the mixture one sees and species may vary considerably over short distances.

Data Describing Mercury Speciation in Soil and Sediment Near Y-12

Three studies between 1984 and 1994, using different extraction and analytical methods, attempted to determine the mercury species in EFPC floodplain soils (Revis et al. 1989, SAIC 1994b). Data from these studies (described in detail in Appendix K) are summarized below:

- Revis et al. (1989) conducted the first study in 1984 using a sequential extraction method. Results implied that most of the mercury (range 84 to 98%) was insoluble mercuric sulfide. Only a minor percentage (0.003 to 0.01%) was identified as methylmercury.

- The USEPA Environmental Monitoring Systems Laboratory (EMSL) conducted the second study in 1993 using a different sequential extraction method (SAIC 1994b) and a different set of soil samples. Results implied that the mercury was a mixture of soluble, insoluble, and elemental mercury in fairly equal proportions. Less than 0.01% was methylmercury.

- USEPA conducted a third study in 1994 using both the Revis et al. (1989) and EMSL methods, as well as an extraction method developed by Sakamoto et al. (1992) (SAIC 1994b). The same set of soil samples was used for all three methods. Reported percentages of elemental mercury and different inorganic mercury species varied considerably. For example, the reported percentage of mercury present as mercuric sulfide ranged from 1 to 105%. The methods also demonstrated poor specificity in extraction of spiked samples.

The wide variation in reported soil speciation was likely due to analytical difficulties, including:

- Wet chemical manipulation techniques (such as leaching and digestion), that are the basis for most extraction techniques, can alter the chemical equilibrium of the system, making the observed speciation an arbitrary function of the extraction conditions (Davis et al. 1996);

- Different species may be extracted by more than one type of chemical extractant, resulting in poor specificity (Davis et al. 1996); and,

- High temperature ovens used to dry soils in some methodologies may alter speciation (Davis et al. 1996).

Because of unresolved difficulties in speciating mercury in soil using current methods, attendees at the USEPA Workshop on Mercury Speciation (Denver, CO, September 1996) concluded that inorganic mercury species in soil will probably never be quantified with certainty using these methods. However, while the percentage of mecury present as different species cannot be definitively quantified, results of the studies of EFPC floodplain soils suggest:

- Concentrations of methylmercury in floodplain soil are very low (<0.01%). Similar findings have been observed at other sites (Rissanen 1975, Lindqvist et al. 1991).

- Forms of mercury soluble in water or weak acids (such as mercuric chloride or mercuric oxide) comprise from 2 to 70% of the total mercury in floodplain soil.

- Insoluble forms of mercury (such as mercuric sulfide) are definitely present in floodplain soil, as confirmed by spectrophotometric techniques. However, the exact percentage measured varied among samples and among extraction and analytical methods.

Based on this information, the project team assumed that mercury in EFPC floodplain soil was a mixture of soluble and insoluble inorganic mercury species and elemental mercury.

While efforts to date have not been successful in quantifying the species of mercury in soil, for purposes of evaluating health risks associated with soil contact, precise quantification of species may be less relevant than understanding the fraction of mercury that is bioavailable in soil, regardless of speciation, relative to the fraction that is bioavailable in the toxicity studies to which soil contact is compared. The fraction of mercury in a given medium that is bioavailable can be considered to be the product of two components: (1) the fraction that is *bioaccessible* (that is, the fraction that desorbs from its matrix under physiological conditions and is available for absorption into the bloodstream), and (2) the portion of the bioaccessible fraction that is actually absorbed into the bloodstream. While studies quantifying the *in vivo* bioavailability (in living systems) of mercury in soil have not been conducted, several *in vitro* (laboratory) methods have

been developed to approximate metal bioavailability in soil by determining its bioaccessibility under conditions simulating gastrointestinal digestion (Ruby et al. 1993, 1996; Paustenbach et al. 1997).

Barnett and Turner of the ORNL Environmental Sciences Division conducted a study to assess the bioaccessibility of mercury in soil using samples from the EFPC floodplain**S** these were the same samples evaluated in the 1994 USEPA soil speciation study (Barnett and Turner 1995). Two samples from each of 10 floodplain locations were digested for four hours in a hydrochloric acid solution of pH 2.5, followed by four hours in a solution of pH 6.5. Study results suggested that the solubility of mercury in most floodplain soil during digestion (based on these simulated conditions) is minimal. Total soluble mercury ranged from 0.30 to 14% in 19 of 20 samples, and 46% in one sample. The average total soluble mercury was 5.3% (see Appendix K for individual sample results). In contrast, 100% of mercuric chloride, the species that is the basis for the regulatory criteria used to evaluate potential health effects from ingestion of inorganic mercury (including the USEPA RfD and the ATSDR MRL), dissolved under these conditions.

Selection of Toxicity Benchmarks and Relative Bioavailability Factors for Evaluating Exposures to Mercury in Soil/Sediment

As indicated in Table 5-10 and described in detail in Section 11.0, most studies of inorganic mercury toxicity following ingestion evaluate highly soluble inorganic mercury salts dissolved in water. Doses from ingestion of mercury soil or sediment were therefore compared to toxicity benchmarks based on ingestion of inorganic mercury. The relative bioavailability PDF for evaluating ingestion of mercury in soil or sediment ($B_{oral\text{-}soil}$) was developed using the data from the relative bioaccessibility assay (Barnett and Turner 1995). This study indicates that the *bioaccessibility* of mercury in soil is low relative to the bioaccessibility of the inorganic mercury species used in the toxicity studies for ingestion of inorganic mercury (average relative bioaccessibility 5.3%). It is assumed that the relationship of the *bioavailability* of mercury in soil relative to the bioavailability of mercuric chloride is the same as the relationship between the bioaccessibility of mercury in soil relative to the bioaccessibility of mercuric chloride.

This study suggests for a given point of contact with EFPC floodplain soil, the most likely approximation of the relative bioavailability of mercury in soil is 5.3%. To address possible spatial differences in the bioavailability of soil mercury at different floodplain locations and depths, results from all 20 of the soil samples analyzed in the assay (relative bioavailability range 0.3% to 46%) were used. Using these date, the relative bioavailability PDF for ingestion of mercury in soil or sediment ($B_{oral\text{-}soil}$) was characterized by a lognormal distribution with a mean of 0.053 and a standard deviation of 0.10.

It is likely that mercury concentrations in surface soil in the floodplain were highest during the period when releases from Y-12 were highest. Later surface soil concentrations were probably lower due to removal of soil during flood events and deposition of suspended materials with lower concentrations. It is assumed, however, that historical relative bioavailability fell within the range of values reflected in the PDF.

5.2.4 Mercury Speciation in Plants

Plants may take up mercury from both soil and air. Overall, plant uptake of mercury from soil through the roots is minimal (Beauford and Barringer 1977; de Temmerman et al. 1986; Mosbaek et al. 1988; Lodenius 1990; Lindberg et al. 1995), due in part to the generally limited solubility of the metal associated with the solid phase in soil (Cataldo and Wildung 1978). Mercury in plants is probably largely due to foliar uptake (through the leaves) of airborne mercury.

Data Describing Mercury Speciation in Vegetation Near Y-12

A few studies have investigated mercury concentrations in vegetables and/or pasture grown in or near the EFPC floodplain (Hibbitts 1986; Van Winkle et al. 1984; SAIC 1994b). None of these studies differentiated between mercury species– results were reported as total mercury (TDHE 1983; Hibbitts 1984; Hibbitts 1986; Van Winkle et al. 1984; SAIC 1994b).

Data on mercury in plants at other sites indicate:

- Speciation studies of mercury in plants grown on soils with elevated inorganic mercury concentrations suggest that, while uptake of mercury from soil is limited, mercury that is taken into the plants is taken up as mercuric ions (i.e., inorganic mercury) (Bache et al. 1973). Although methylmercury was detected in plants at concentrations up to 36% of the total mercury when plants were grown on soils amended with methylmercury or with sewage sludge, these studies suggest it is unlikely that methylmercury is formed within plants grown in inorganic mercury-contaminated soil under field conditions (Bache et al. 1973; Cappon 1981; Fortmann et al. 1977).

- Plants may take up mercury from air by foliar absorption of elemental mercury or uptake of soluble divalent mercury following deposition on the plant surface. In an area with high airborne mercury concentrations, Mosbaek et al. (1988) estimated that 90% of the total plant mercury in the "green parts" of the plant was contributed by airborne mercury. Inside the plant, absorbed mercury equilibrates between Hg^0 (elemental mercury), Hg^+ (monovalent mercury), and Hg^{2+} (divalent mercury).

Based on this information, the project team assumed that mercury in vegetation near the ORR was a mixture of inorganic and elemental mercury species.

Selection of Toxicity Benchmarks and Relative Bioavailability Factors for Evaluating Exposures to Mercury in Vegetation

As indicated in Table 5-10 and described in detail in Section 11.0, most studies of inorganic mercury toxicity following ingestion evaluate highly soluble inorganic mercury salts dissolved in water. Doses from ingestion of mercury in vegetation were therefore compared to toxicity benchmarks based on ingestion of inorganic mercury. It is expected that the presence of fibrous material in ingested vegetation may have some effect on limiting the bioavailability of ingested mercury. Similar observations have been made of the effect of soil on reducing the bioavailability of ingested lead compared to lead ingested without soil**S** this was hypothesized to be due in part to the soil acting as a fiber source, absorbing the lead in the gastrointestinal tract and reducing net absorption (Chaney et al. 1989). Although limited data are available, the bioavailability of elemental mercury following ingestion is considered to be less than soluble inorganic mercury compounds such as mercuric chloride (ATSDR 1997).

Based on these considerations, doses from ingestion of mercury in vegetation were evaluated assuming that the bioavailability of mercury in vegetation may have ranged from somewhat less than to as bioavailable as the mercury in the toxicity studies. The relative bioavailability PDF for evaluating ingestion of mercury in vegetation ($B_{oral\text{-}veg}$) was characterized by a uniform distribution with a minimum of 0.6 and a maximum of 1.0.

5.2.5 Mercury Speciation in Fish

Studies of mercury speciation in fish downstream of the Y-12 Plant and at other sites have shown that mercury is predominantly present in fish muscle as methylmercury.

Data Describing Mercury Speciation in Fish Near Y-12

Data on mercury in fish downstream from the ORR and at other sites indicate:

- 94.8 ± 10.7 % (range 65 to 103%) of mercury in the muscle of ten fish collected in Poplar Creek in 1974 was in the methyl form (Elwood 1977).

- Studies at other sites, both freshwater and saltwater, agree that most mercury in fish (typically >95%) is methylmercury (Bishop and Neary 1974, Westoo 1973, Zitko et al. 1971, Bloom 1992).

- The proportion of methylmercury is independent of fish size, length, or weight, with no significant differences in percent methylmercury between species (Bishop and Neary 1974).

Based on these data, the project team assumed that mercury in fish near the ORR was predominantly methylmercury.

Selection of Toxicity Benchmarks and Relative Bioavailability Factors for Evaluating Exposures to Mercury in Fish

As indicated in Table 5-10 and described in detail in Section 11.0, studies of methylmercury toxicity following ingestion evaluate human populations that ingested methylmercury in feed grain or in fish. While the percent of mercury in fish that is methylmercury varies among individual fish (between approximately 90 and 100%), it is assumed that the distribution of percent methylmercury in freshwater fish caught near the ORR was the same as in the mostly saltwater fish that are the basis for recent studies of the adverse health effects from ingestion of mercury in fish.

Based on these considerations, doses associated with ingestion of mercury in fish were estimated assuming that the bioavailability of mercury in fish is the same as in the toxicity studies (i.e., the *relative* bioavailability is 1.0). The relative bioavailability PDF for evaluating ingestion of mercury in fish ($B_{oral\text{-}fish}$) was characterized by a point estimate with a value of 1.0.

5.2.6 Summary of the Mercury Species Assumed to be Present in Each Medium

The species of mercury assumed in this assessment to be present in each exposure medium, and the relative bioavailability PDF assumed for exposure to mercury in each medium, are summarized in Table 5-11.

Table 5-11: Speciation Assumptions for Each Exposure Medium

Exposure Medium	Mercury Species Assumed to be Present	Basis for Reference Toxicity Benchmark Values	Relative Bioavailability PDF
Air	Elemental mercury vapor	Inhalation exposure of adult workers to elemental mercury vapor	$B_{inhalation}$ = 1.0 (point estimate)
Surface Water	Mixture of soluble and insoluble inorganic mercury species and insoluble inorganic mercury sorbed to particulates	Ingestion of soluble mercuric chloride by rodents	$B_{oral\text{-}water}$ = 0.8 to 1.0 (uniform distribution)
Soil/Sediment	Mixture of soluble and insoluble inorganic and elemental mercury species	Ingestion of soluble mercuric chloride by rodents	$B_{oral\text{-}soil}$ = 0.053 (mean) (lognormal dist., SD = 0.1)
Plants	Mixture of soluble and insoluble inorganic and elemental mercury species	Ingestion of soluble mercuric chloride by rodents	$B_{oral\text{-}veg}$ = 0.6 to 1.0 (uniform distribution)
Fish	Methylmercury	Ingestion of methylmercury in fish or treated grain	$B_{oral\text{-}fish}$ = 1.0 (point estimate)

6.0 IDENTIFICATION AND CHARACTERIZATION OF EXPOSURE PATHWAYS AND POTENTIALLY EXPOSED POPULATIONS

This section describes the exposure pathways and populations evaluated in the assessment of exposures to mercury released from the ORR. Steps in the identification and characterization of exposure pathways and potentially exposed populations include:

- **Identifying complete exposure pathways** through which individuals may have been exposed to mercury in the environment;

- **Identifying and characterizing exposed populations**, including their location and the pathways through which these populations may have been exposed to mercury, and characterizing potentially sensitive population subgroups; and

- **Identifying appropriate equations to describe exposure** via each pathway identified as potentially complete.

For each exposure population and exposure pathway, doses were estimated using a derivation of the dose equation described in Section 2.1, combined with population- and pathway-specific exposure point concentrations (described in Section 7.0), transfer factors (described in Section 8.0), and exposure parameters (described in Section 9.0).

6.1 Identification of Complete Exposure Pathways

The presence of a contaminant in the environment does not necessarily mean that exposure will occur. For exposure to occur, a pathway of exposure through which an individual contacts and takes up the contaminant from the environment must exist. Several factors, including environmental conditions, the potential for a chemical to move from one medium to another, and the lifestyles and characteristics of the potentially exposed population, can influence whether a pathway is complete.

For mercury historically used at Y-12 to have posed a health hazard to off-site individuals, each of the following elements of a complete exposure pathway must have existed:

- A **source** that released mercury to the environment,

- A **transport medium** that carried the mercury off-site to a location where exposure could have taken place, and

- An **exposure route** through which mercury entered an individual's body.

If any of these elements was missing, the pathway was not complete.

Based on the above criteria, a number of pathways of exposure to mercury historically released from the ORR are likely to have been complete for some individuals in nearby populations. Potential exposure pathways to mercury in different environmental media are described in the following sections.

6.1.1 Pathways Associated with Exposure to Mercury in Surface Water

As described in Sections 3 and 4, between 1953 and 1962, large quantities of mercury were released directly from Y-12 to EFPC. After cessation of the Colex process in 1962, mercury releases to EFPC were much lower, although some mercury continued to be released from Y-12. After 1963, much of this mercury was retained in New Hope Pond, constructed as an equalization basin for upper EFPC surface water exiting Y-12. Several residents who historically lived near the creek report that in the late 1940s and early 1950s, the creek often had a foul smell and occasionally a milky color (DaMassa 1995).

In recent years (after 1982), the Tennessee Department of Health and Environment (TDHE) has posted EFPC with signs advising against contact with water or ingestion of fish from EFPC. However, in most areas of EFPC, there are no physical restrictions to access. Further, individuals who historically resided near EFPC indicated that, during the 1950s and 1960s, they occasionally contacted EFPC surface water during farming or recreational activities. Some former residents of Oak Ridge report having played in EFPC and nearby creeks as children for periods of up to eight hours per day several times per week during the summer (DaMassa 1995). In addition, individuals who lived near EFPC reported that livestock that grazed along EFPC ingested surface water from the creek, suggesting that exposure to mercury in milk or meat from these animals may have occurred. Interviews with individuals who historically farmed or raised vegetables adjacent to EFPC indicate that EFPC surface water was not used for irrigation (DaMassa 1995).

Data collected downstream from Y-12 from 1955 to 1961 show that, compared to concentrations in EFPC, surface water concentrations in downstream waterways (including Poplar Creek, the Clinch River, and Watts Bar Reservoir) were 7 to 350 fold lower (Kwasnoski and Whitson 1955-1961) (Table 6-1). These lower concentrations were likely the result of a combination of factors, including dilution, volatilization, and adherence of mercury to particulates followed by settling out of solution. Because of this reduction in downstream concentrations, exposures to mercury in surface water (not including fish ingestion) were evaluated in the current assessment only for populations exposed to water in EFPC, not those who may have contacted water further downstream.

Pathways of exposure to mercury in EFPC surface water considered in this assessment are listed below.

Table 6-1: Concentrations of Mercury in Surface Water Downstream from Y-12 to EFPC

Study and Year of Sample Collection	Location [a,b,c]	Mean Concentration ($mg\ L^{-1}$)
Kwasnoski and Whitson (1955-61)		
1955	Y-12 (Daily effluent at weir)	1.67
1956	EFPC/ Poplar Creek junction (~EFPC Mile 0.5)	0.147
1957	EFPC/ Poplar Creek junction (~EFPC Mile 0.5)	0.191
1958	EFPC/ Poplar Creek junction (~EFPC Mile 0.5)	0.102
1959	EFPC/ Poplar Creek junction (~EFPC Mile 0.5)	0.014
1960	EFPC/ Poplar Creek junction (~EFPC Mile 0.5)	0.011
1961	EFPC/ Poplar Creek junction (~EFPC Mile 0.5)	0.0097
1955-1957	Poplar Creek (~PCM 4.5)	0.055
1955-1957	Clinch River (~CRM 14.5)	0.0048
1956-1957	Clinch River (~CRM 10.5)	0.0049
Sanders (1970)		
1970	EFPC (New Hope Pond)	0.00038
1970	EFPC (~EFPCM 14)	0.0002
1970	EFPC (~EFPCM 9.7)	<0.0002
1970	EFPC (~EFPCM 6.5)	<0.0002
UCC (1976-1983)		
1975-1982	Clinch River (~CRM 11)	<0.001
1975-1982	Clinch River (~CRM 9.7)	<0.001
TVA (1985a)		
1984	EFPC (EFPCM 14.36)	0.0014
1984	Poplar Creek (PCM 13.8)	<0.0002
1984	Clinch River (CRM 24.0)	<0.0002
1984	Clinch River (CRM 6.8)	<0.0002
MMES (1988-1992)		
1987, 1989, 1991	Clinch River (~CRM 9.7)	<0.0002
1989, 1991	Clinch River (~CRM 10.5)	0.00017
Olsen et al. (1990)		
1989	Watts Bar Reservoir (Whites Creek)	0.00021
Cook et al. (1992)		
1990	Poplar Creek (PCM 0 to 5.5)	0.00028
1990	Clinch River (CRM 0 to 12)	<0.0002
1990	Watts Bar Reservoir	<0.0002

a For each study, locations are listed in order of increasing downstream distance from Y-12 effluent release point (EFPC Mile 14.36)
b EFPCM = East Fork Poplar Creek Mile; PCM = Poplar Creek Mile; CRM = Clinch River Mile
c The confluence of EFPC with Poplar Creek is at approximately PCM 5.5. The confluence of Poplar Creek with the Clinch River is at approximately CRM 12.

Exposure Pathways to Mercury in Surface Water

Surface water 6 Humans (Incidental ingestion)
Surface water 6 Humans (Dermal contact)
Surface water 6 Livestock (Meat) 6 Humans
Surface water 6 Dairy cattle (Milk) 6 Humans

The methods used by the project team to estimate historical mercury concentrations in surface water that may have been contacted by Oak Ridge residents or ingested by livestock are discussed in Section 7.1.

6.1.2 Pathways Associated with Exposure to Mercury in Air

As described in Sections 3 and 4, between 1953 and 1962, large quantities of mercury were released directly to air from Y-12 primarily as a result of building ventilation systems installed to lower concentrations of airborne mercury inhaled by workers in the lithium isotope separations facilities. It is likely that mercury also volatilized from EFPC and from soil in the EFPC floodplain. Airborne mercury may have been inhaled by humans and/or livestock, or deposited on or absorbed by vegetation and subsequently ingested by humans and/or livestock. However, uptake of mercury by livestock through direct inhalation of airborne mercury is likely to have been minor compared to other routes of uptake (such as ingestion of mercury in surface water or in pasture grass). Therefore, the direct inhalation pathway was not evaluated for livestock.

Exposure pathways evaluated for mercury in air are listed below.

Exposure Pathways to Mercury in Air Releases

Air 6 Humans (Inhalation)
Air 6 Above-ground Fruits and Vegetables 6 Humans
Air 6 Pasture 6 Livestock (Meat) 6 Humans
Air 6 Pasture 6 Dairy cattle (Milk) 6 Humans

The methods used by the project team to estimate historical mercury concentrations in air that may have been inhaled by nearby populations or taken up by vegetation are discussed in Sections 7.2 and 7.3.

6.1.3 Pathways Associated with Exposures to Mercury in Soil/Sediment

Some of the mercury released to air and surface water was deposited on soil or sediment near the ORR. Sampling data indicate that mercury concentrations in soil in the EFPC floodplain are elevated. Further, core samples of floodplain soil show stratification of mercury concentrations in the soil, with the highest concentrations typically at depths of approximately six-inches to one-foot below the surface (SAIC 1994b). These data suggest mercury concentrations in surface soil were higher in the past, peaking during the years of highest releases from Y-12, and, during subsequent years, these soils were overlain with less contaminated material. In general, areas of highest concentrations are in areas of the floodplain most frequently inundated by flood waters.

Mercury occurs naturally in soils (Henke et al. 1993). Background concentrations of mercury in soils in the Eastern United States range from <0.01 to 3.4 mg kg^{-1} (Dragun and Chiasson 1991, see Table 5-8). Mercury concentrations measured by Oak Ridge Associated Universities (ORAU) in soil samples collected in the city of Oak Ridge outside of the floodplain between 1983 and 1987 were typically less than 1 mg kg^{-1} (TDHE 1983; Hibbitts 1984; Hibbitts 1986). Therefore, in this assessment, the project team focused on evaluating exposures to mercury in soil and sediment in the EFPC floodplain. Exposures to soil mercury may have occurred through direct contact with soil or sediment, root uptake of mercury from soil into vegetables and subsequent vegetable ingestion, and/or root uptake or surface contamination of pasture with soil mercury and ingestion of beef and/or milk from cattle that grazed on the pasture.

Exposure pathways evaluated for mercury in soil or sediment are listed below.

Exposure Pathways to Mercury in Soil or Sediment

Soil/ Sediment 6 Humans (Incidental ingestion)
Soil/ Sediment 6 Humans (Dermal contact)
Soil 6 Below-ground Vegetables 6 Humans
Soil 6 Livestock (Meat) 6 Humans
Soil 6 Dairy cattle (Milk) 6 Humans
Soil 6 Pasture 6 Livestock (Meat) 6 Humans
Soil 6 Pasture 6 Dairy cattle (Milk) 6 Humans

The methods used by the project team to estimate historical mercury concentrations in soil/sediment that may have been contacted by Oak Ridge residents or ingested by livestock are discussed in Section 7.4.

6.1.4 Fish Ingestion

Inorganic mercury in surface water systems may be converted to methylmercury by microorganisms in the water column or bottom sediments, although concentrations of methylmercury in water or sediment are generally very low (Huckebee et al. 1975; Gilmour et al. 1992). Fish and other aquatic species may absorb or bioconcentrate methylmercury either directly through the water or through the components of the food chain. Because the rate of uptake of methylmercury by fish is rapid and the clearance rate very slow, the net result is high methylmercury concentrations in fish relative to surrounding water and sediments (Huckabee et al. 1975; Henke 1993; Bogle 1972; Cole et al. 1992). A number of studies show a correlation between concentrations of mercury in fish and concentrations in surface water or sediment. In the absence of direct measurements in fish, these correlations have been used to predict fish concentrations.

While individuals may have occasionally caught and consumed fish from EFPC, it is unlikely that the numbers of fish in the creek during the years of peak releases (i.e., early 1950s through mid-1960s) were sufficient to support subsistence fishing (Barnthouse and Deppen 1996). However, fish populations were significantly larger further downstream in Poplar Creek, the Clinch River, and Watts Bar Reservoir.

The methods used by the project team to estimate historical mercury concentrations in fish are discussed in Section 7.5.

6.2 Selection and Characterization of Exposed Populations

The dose an individual receives depends greatly on *where* the individual is exposed. For example, as airborne contaminants are transported away from the release point, air concentrations decrease through dilution and deposition onto the ground or on other objects. In addition, individuals residing at different locations have different day-to-day activity patterns that affect the rate at which they may be exposed to a contaminant.

Individuals who historically lived near the ORR could have been exposed to mercury released from the ORR at a number of locations. In this assessment, populations at several locations were investigated to capture potential variations in population- or site-specific intake rates and exposure point concentrations. These "reference" populations include those likely to have had the highest exposures, due to their activity patterns and their proximity to release points and areas of high off-site concentrations, and those with lower or more typical exposures reflective of larger segments of the population.

This section describes the reference populations considered in this assessment and the exposure pathways through which it was assumed they may have been historically exposed to mercury released from the ORR. Locations of these populations are shown in Figure 1-2. Although exposure characteristics and exposure point concentrations for specific individuals in these populations may vary from those modeled in this assessment, the population groups selected are intended to reflect exposures to a range of individuals who resided in the Oak Ridge area.

6.2.1 Selection and Characterization of Reference Population Groups

As described in Section 1, air exchange between Y-12 and the city of Oak Ridge is largely impeded by Pine Ridge, and winds blow predominantly northeast and southwest along Bear Valley, following the local terrain. However, EFPC flows through a gap in Pine Ridge, beyond New Hope Pond/ Lake Reality, and flows through business and residential areas in the city of Oak Ridge before joining Poplar Creek about 14 miles downstream from Y-12. Thus, individuals exposed to mercury released from Y-12 likely included people living downvalley from Y-12, exposed directly to mercury in air releases, as well as individuals living on the other side of Pine Ridge, exposed to mercury in EFPC. In addition, people who fished in waterways further downstream, including Poplar Creek, the Clinch River, and Watts Bar Reservoir, may have been exposed to methylmercury in fish.

Based on the locations and activity patterns of people who historically lived near the Y-12 Plant, several populations downvalley or downstream of Y-12 were selected for evaluation in the dose reconstruction. These population groups were selected to estimate exposures to:

(1) *Residents who lived in closest proximity to the Y-12 facility*. These included residents who lived in Wolf Valley, the area of nearest downwind residences from Y-12, and residents of the Scarboro Community, who lived less than one-half mile from Y-12, on the opposite side of Pine Ridge.

(2) *Residents who were likely among the most highly exposed off-site individuals*, due primarily to their activity patterns as well as the location of their residences. These included residents who lived in small rural farms at the western end of the city of Oak Ridge, directly adjacent to the EFPC floodplain, and who were known to have engaged in backyard gardening and raising livestock for personal use.

(3) *Residents who lived further from EFPC in suburban settings typical of the lifestyle of most Oak Ridge residents*, but who may have been exposed to airborne mercury that volatilized from the creek. While exposures to individuals in these population groups were likely lower than some of the other groups, it is assumed that the size of these population groups was substantially larger.

(4) *Students who attended a school directly adjacent to EFPC* (Robertsville Junior High School), who may have been exposed to airborne mercury that volatilized from the creek, or played in and around the creek.

(5) *Individuals who fished in downstream waterways*, including Poplar Creek, the Clinch River, and Watts Bar Reservoir, and may have been exposed to mercury in fish that they consumed.

Population groups evaluated in this assessment are described below. For each population group, exposures parameters PDFs were defined to characterize the range of exposures likely within the group of individuals making up the population. Much of the information used to describe behaviors and possible exposures of these population groups was gathered during interviews with area residents (compiled in DaMassa 1995) or from historical literature and land use information. Other information (including intake rates) is based largely on published information in the scientific literature that statistically evaluates exposure patterns in different populations in the United States (including data from the United States Environmental Protection Agency (USEPA) and the United States Department of Agriculture (USDA)). To some extent, however, there is uncertainty about the true value of the parameters used to describe exposure (such as rates describing how much milk or vegetables people consumed, or how often they played in EFPC). Consequently, exposure parameter PDFs were also selected to encompass the uncertainty about the true value of a parameter for individuals in the population. Exposure parameter PDFs for each population group are summarized in Section 9 and described in detail in Appendix V.

6.2.1.1 Wolf Valley Residents

Activities and Characteristics of the Population

Meteorological data collected between 1987 and 1992 by X-10 personnel at a tower near the east end of the Y-12 Plant show that winds blow predominantly northeast and southwest (Figure 6-1) along Bear Creek/ Union Valley between Pine Ridge (approximately 300 feet high) and Chestnut Ridge, following the local terrain. The nearest dwelling historically present along the predominant airflow direction during the time of maximum airborne releases (1953 to 1962) was on the opposite side of the Clinch River (now Melton Hill Lake), in the extension of Union Valley about five miles northeast of the Y-12 Plant (DaMassa 1995). This rural setting, known as the Wolf Valley area, has consisted of residences and small farms since before World War II (DaMassa 1995).

In this assessment, it was assumed that individuals (adults and children) residing in Wolf Valley raised garden fruits (e.g., berries) and vegetables, dairy cows, and beef cattle, and may have been exposed to mercury through the pathways listed below.

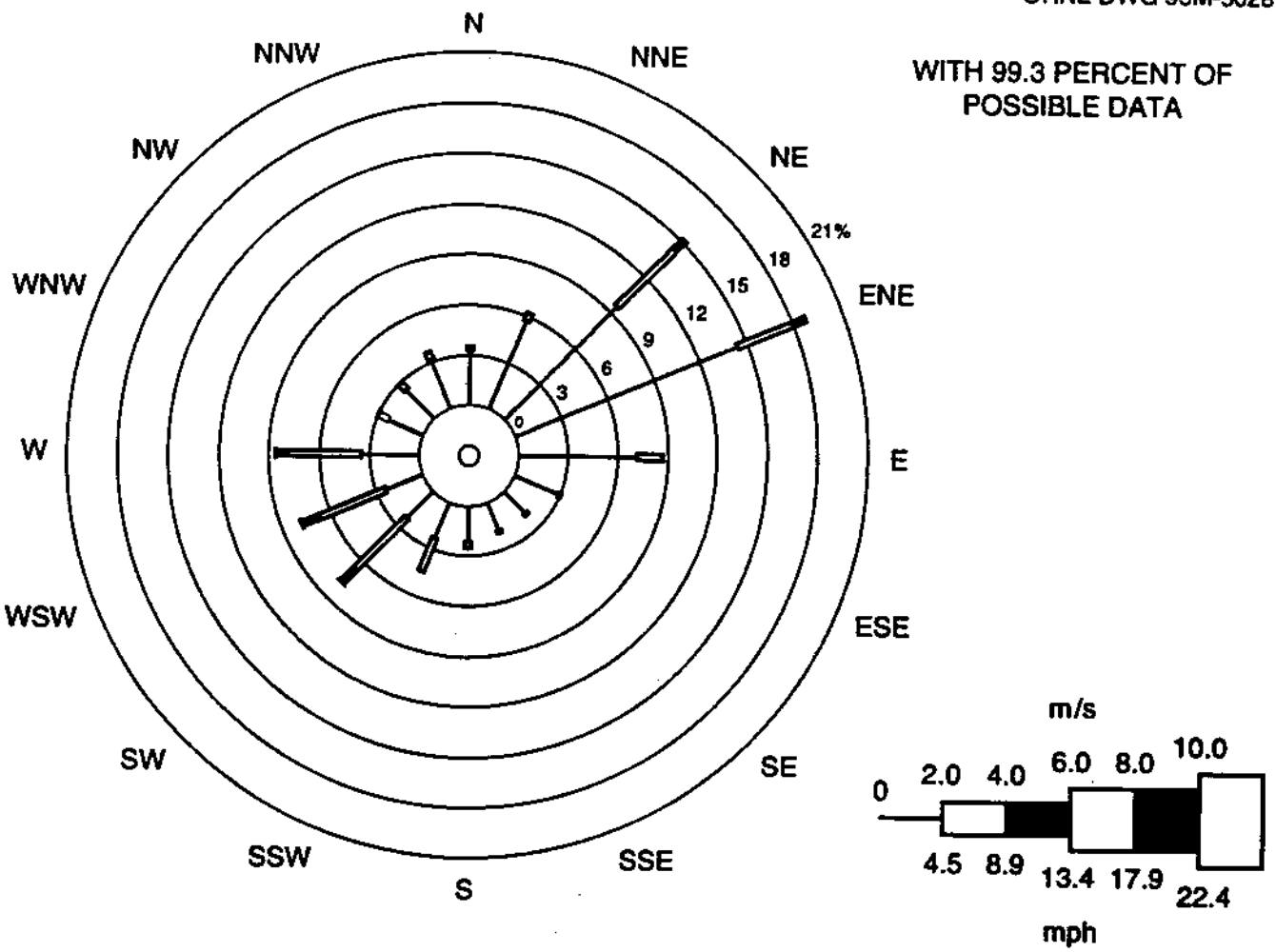


Figure 6-1: Wind rose for the Y-12 facility [based on measurements from meteorological tower "MT5" (east), 1992, 10-m level].

NOTE: A wind rose graphically depicts frequencies of wind directions and speeds at a location where measurements have been made. There are 16 lines or "spokes" radiating out from the center of the wind rose, one for each 22.5-degree wind direction sector (i.e., N, NNE, NE, etc.). Wind roses are typically prepared so that the north sector represents true north (as opposed to magnetic north or a locally-defined grid north). The length of the line in each sector indicates the fraction of total hours that the wind was blowing *from* that stated direction. In this example, each concentric ring that a line crosses represents 3 percent of the measured data. For example, the wind was blowing from the northeast just over 15 percent of the time. The prevalences of different wind speed classes are often depicted by varying widths along the line in each sector. In this wind rose, for example, a majority (approximately 75%) of the winds from the ENE fell within the 0 to 2.0 meters per second wind speed class, while about 20% fell within the 2.0 to 4.0 m/s class and about 5% fell in the 4.0 to 6.0 m/s class.

Exposure Pathways Evaluated for Wolf Valley Residents

Air *(pathways associated with direct airborne releases from Y-12)*
Air ÷ Humans (Inhalation)
Air ÷ Above-ground Fruits and Vegetables ÷ Humans (Ingestion)
Air ÷ Pasture ÷ Dairy cows (Milk) ÷ Humans (Ingestion)
Air ÷ Pasture ÷ Livestock (Meat) ÷ Humans (Ingestion)

Population Size

Examination of USGS quadrangle maps (Clinton quadrangle) for 1952, 1968, 1975, and 1990 shows that the number of buildings in the Wolf Valley area within one mile of Clinch River/ Melton Hill Lake increased slightly between 1952 and 1990 from about 14 to 20 buildings (USGS 1952, 1968b, 1975, 1990). Based on this number of residences, it is estimated that during a given year between 1950 and 1990, the size of the Wolf Valley population within one mile of Clinch River/ Melton Hill Lake was between 30 and 100 individuals. Per Israeli and Nelson (1992), the average residence time for a farm family household (based on data from 1985 and 1987 U.S. housing surveys) was 17.3 years; individuals within a given household may have lived at the residence for a shorter time. Assuming that in any given year, 1/15 of the Wolf Valley population left and was replaced by new residents, the total population size between 1950 and 1990 was estimated to be between 100 and 350 individuals.

6.2.1.2 Scarboro Community Residents

Activities and Characteristics of the Population

The Scarboro Community is located approximately 0.3 miles north of Y-12 on the opposite side of Pine Ridge. The Scarboro Community was built in the late 1940s to house black workers who had been living in hutments in other areas of Oak Ridge. The decision was made to establish the Scarboro Community at the former site of the Gamble Valley Trailer Camp in 1948. The Scarboro Community was first opened to residents in 1950, and has been continuously inhabited since.

The Scarboro Community is a residential area consisting of single family homes. Some families had small backyard gardens. Although meteorological studies indicate that winds near the Y-12 Plant predominantly follow Bear Creek/ Union Valley, meteorological studies and ambient air monitoring programs (including an air monitoring program for uranium from the Y-12 Plant) indicate that the local ridges are not perfect barriers– some degree of transport of airborne effluents from the Y-12 Plant into adjacent valleys occurs (the elevation of the Y-12 building exhaust was as much as 90 feet above the ground, compared to the elevation of Pine Ridge 300 feet above the ground). Low concentrations of mercury have been measured in soils around the Scarboro Community, resulting most likely from deposition of airborne mercury from either direct releases from Y-12 or volatilization of mercury from EFPC.

In addition to exposures to mercury while at home, some residents of the Scarboro Community reported that they traveled the short distance to EFPC to engage in recreational fishing or other activities such as wading or playing in the creek, particularly during the 1950s when the city of Oak Ridge was segregated and access to some recreational facilities was restricted. Fish and other aquatic biota reportedly caught during the 1950s and 1960s by Scarboro residents included bluegill, stripers, crappies, bass, sunfish, crayfish, and turtles (DaMassa 1995). Accounts differ as to whether water quality in EFPC during this period could have supported fish populations (DaMassa 1995)**S** it has been suggested that fish reportedly caught in EFPC were actually caught in tributaries to EFPC; however, for purposes of this assessment, it is assumed that individuals who lived in the Scarboro Community and fished in EFPC occasionally consumed fish that they caught from the creek.

In this assessment, it was assumed that individuals (adults and children) residing in the Scarboro Community could have been exposed to mercury through the pathways listed below.

Exposure Pathways Evaluated for Scarboro Community Residents

Air *(pathways associated with both direct airborne releases from Y-12 and volatilization*
Air ÷ Humans (Inhalation)
Air ÷ Above-ground Fruits and Vegetables ÷ Humans (Ingestion)
Soil *(pathways associated with backyard soil)*
Soil ÷ Humans (Incidental ingestion)
Soil ÷ Humans (Dermal contact)
Soil ÷ Below-ground Vegetables ÷ Humans (Ingestion)
Sediment *(pathways associated with EFPC sediment)*
Sediment ÷ Humans (Incidental ingestion)
Sediment ÷ Humans (Dermal contact)
Surface Water *(pathways associated with EFPC water)*
Surface water ÷ Humans (Incidental ingestion)
Surface water ÷ Humans (Dermal contact)
Fish *(pathway associated with EFPC fish)*
Fish ÷ Humans (Ingestion)

Population Size

The 1950 census indicated that the non-white population of Oak Ridge was approximately 1,200 people. USGS quadrangle maps (Bethel Valley Quadrangle) for 1953 shows approximately 200 houses in the Scarboro area (USGS 1953a). A number of additional buildings (about 10 to 20) are apparent in the 1968 and 1989 quadrangles (USGS 1968a, 1989). Based on the census data and USGS maps, it is estimated that the population of Scarboro during a given year between 1950 and 1990 was between 800 and 1,200 people. Per Israeli and Nelson (1992), the average residence times for rural and urban households (based on data from 1985 and 1987 U.S. housing surveys) were 7.8 and 4.2 years, respectively. Assuming that the average residence time for individuals in the Scarboro Community was intermediate between these values (about 6 years), it was assumed that in any given year, 1/6 of the population may have left and been replaced by new residents. Based on these assumptions, the total population size between 1950 and 1990 was estimated to be between 6,000 and 10,000 individuals.

Given the size and characteristics of EFPC and its low productivity, it is unlikely that a substantial number of anglers used it as a fishery. Anecdotal information indicates that EFPC has not always supported viable fish populations as a result of industrial releases from Y-12. Because of these factors, the project team assumed that the total population of anglers in EFPC between 1950 and 1990 was less than 100 individuals.

6.2.1.3 Robertsville School Children

Activities and Characteristics of the Population

Robertsville Junior High School is located on Robertsville Road, north of the Oak Ridge Turnpike and just west of Illinois Avenue. The schoolyard is directly adjacent to EFPC, on the north side of the creek at approximately EFPC Mile 12. The school is located at the site of the original Robertsville School, a country schoolhouse that belonged to the community of Robertsville, one of four original farming communities displaced prior to construction of the ORR. In the early 1940s, Jefferson School was built in temporary buildings at the site, incorporating the original schoolhouse. Initially an elementary school, Jefferson School was converted to a junior high school, comprising grades seven through nine, shortly before the end of World War II to relieve pressure on the overcrowded Oak Ridge High School. In the early 1950s, the temporary buildings were replaced by permanent structures, and the school was renamed Robertsville Junior High School (Skidmore Owings & Merrill 1948, Overholt 1987).

The Robertsville Junior High School yard includes several playfields that directly abut EFPC. Parts of the school yard have flooded during the last 50 years. For example, during an April 1956 flood, overflow from EFPC covered the football field end zone to a depth of approximately one foot (TVA 1959). Historically, there has been a footbridge across EFPC at the location of the school (TVA 1959), and there are no physical restrictions to creek access. In this assessment, it is assumed that Robertsville Junior High School students may have been exposed to airborne mercury and contacted contaminated soils in the school yard, particularly the area closest to the creek that was subject to occasional flooding.

In addition, some students may have occasionally engaged in recreational activity in or along EFPC (such as wading), resulting in exposure to mercury in sediment or surface water in the creek. Some former residents of Oak Ridge report having played in EFPC and nearby creeks as children for periods of up to eight hours per day several times per week during the summer, and that there was a swimming hole used by a small number of children during the 1950s between approximately EFPC Mile 10 and 11, near the present site of the Weigle's convenience store (DaMassa 1995). Two individuals who grew up near EFPC in the late 1940s and 1950s reported that construction workers occasionally gave the children plywood, and the children would build rafts to float down the creek (DaMassa 1995).

In this assessment, it was assumed that Robertsville Junior High School children could have been exposed to mercury through the pathways listed below.

Exposure Pathways Evaluated for Robertsville School Children

Air *(pathway associated with volatilization from EFPC)*
Air ÷ Humans (Inhalation)
Soil *(pathways associated with schoolyard soil)*
Soil ÷ Humans (Incidental ingestion)
Soil ÷ Humans (Dermal contact)
Sediment *(pathways associated with EFPC sediment)*
Sediment ÷ Humans (Incidental ingestion)
Sediment ÷ Humans (Dermal contact)
Surface Water *(pathways associated with EFPC water)*
Surface water ÷ Humans (Incidental ingestion)
Surface water ÷ Humans (Dermal contact)

Population Size

Robertsville Junior High School was built to serve approximately 2,000 students (Skidmore Owings & Merrill 1948). It was assumed that these students were primarily exposed to airborne mercury and mercury in contaminated floodplain soil. Assuming about one-third of these students were new every year, the total population size between 1950 and 1990 was estimated to be between 20,000 and 30,000 students.

It was more difficult to determine the fraction of school-age children who regularly used the stream for recreational purposes and may have been exposed to mercury in surface water and sediment. The attractiveness of EFPC for recreational purposes is limited in many areas due to the varying depth, width and steepness of the banks. In addition, access would have been limited in some areas due to proximity

to private properties and roads. Although a footpath used by schoolchildren to cross the stream was identified, the existence of a bridge along the path was assumed to have limited the likelihood of contact with the sediments and surface water of the stream. Based on these considerations, the project team assumed that the children regularly using the stream were those who lived in close proximity to the stream. Based on interviews with area residents, the project team assumed that approximately 5 to 20 of the elementary school age children who lived near the stream at any one time used it for recreational activities. It was assumed that a new group of children began using the stream every six years, resulting in an estimated total population size between 1950 and 1990 of 100 to 300 children.

6.2.1.4 EFPC Floodplain Farm Family

Activities and Characteristics of the Population

Land within the boundaries of the EFPC floodplain has historically been mostly undeveloped. However, data gathered during interviews of current and former Oak Ridge-area residents indicate that several families lived adjacent to the EFPC floodplain and farmed in or near the floodplain (DaMassa 1995). These families resided between approximately EFPC Miles 8 and 11 (near the present day location of Windsor Drive). Farming activities included raising beef and/or dairy cattle. Typically, when beef cattle were slaughtered, some of the meat was kept for family use and some was sold. Milk from dairy cattle was kept for family use. Individuals who lived near EFPC reported that livestock that grazed along EFPC ingested surface water from the creek.

Farm families who lived near EFPC also grew fruits and vegetables in backyard gardens, some of which was reportedly canned or frozen for year-round use (DaMassa 1995). Interviews with individuals who farmed or raised vegetables adjacent to EFPC indicate that EFPC surface water was not used for irrigation (DaMassa 1995).

Individuals who historically lived near EFPC indicated that, during the 1950s and 1960s, they occasionally contacted EFPC surface water during farming or recreational activities. Members of one farm family report that their children were taught how to swim in EFPC (DaMassa 1995). Members of these families, particularly children, reportedly occasionally caught fish from EFPC; however, it is unlikely that the fish populations during the years of peak releases were sufficient to support subsistence fishing or that consumption of large numbers of fish was desirable due to the apparent contamination of the creek (DaMassa 1995, Barnthouse and Deppen 1996).

In addition to livestock grazed in the floodplain by families who lived directly adjacent to the floodplain, several larger commercial herds were also reported to have been grazed in the floodplain during the 1950s through 1970s (DaMassa 1995). These included a herd on the Oak Ridge Turnpike across from the Bruner's Center from 1961 to 1974 (at approximately EFPC Mile 11), a herd near the inflow of Mill Creek during the 1950s and 1960s (at approximately EFPC Mile 9), and a herd just east of the present day location of the Oak Ridge Country Club during the 1950s and 1960s (at approximately EFPC Mile 8). These were reportedly beef cattle raised for slaughter; these commercial herds reportedly drank water

from EFPC. In addition, after 1965, a herd of beef cattle was reportedly grazed further upstream on EFPC, across the creek from the NOAA site (at approximately EFPC Mile 14). Unlike the other herds, these animals reportedly did not drink water from the creek, but from a spring that originated on the "Y-12 Hill" (DaMassa 1995).

The EFPC floodplain farm family population was assumed to reside at approximately EFPC Mile 10, with potential exposure to mercury through the pathways listed below.

Exposure Pathways Evaluated for the EFPC Floodplain Farm Family

Air *(pathways associated with volatilization from EFPC)*

Air ÷ Humans (Inhalation)

Air ÷ Above-ground Fruits and Vegetables ÷ Humans (Ingestion)

Air ÷ Pasture ÷ Dairy cows (Milk) ÷ Humans (Ingestion)

Air ÷ Pasture ÷ Livestock (Meat) ÷ Humans (Ingestion)

Soil *(pathways associated with EFPC floodplain soil)*

Soil ÷ Humans (Incidental ingestion)

Soil ÷ Humans (Dermal contact)

Soil ÷ Below-ground Vegetables ÷ Humans (Ingestion)

Soil ÷ Livestock (Meat) ÷ Humans (Ingestion)

Soil ÷ Dairy cows (Milk) ÷ Humans (Ingestion)

Soil ÷ Pasture ÷ Livestock (Meat) ÷ Humans (Ingestion)

Soil ÷ Pasture ÷ Dairy cows (Milk) ÷ Humans (Ingestion)

Sediment *(pathways associated with EFPC sediment)*

Sediment ÷ Humans (Incidental ingestion)

Sediment ÷ Humans (Dermal contact)

Surface Water *(pathways associated with EFPC water)*

Surface water ÷ Humans (Incidental ingestion)

Surface water ÷ Humans (Dermal contact)

Surface water ÷ Livestock (Meat) ÷ Humans (Ingestion)

Surface water ÷ Dairy cows (Milk) ÷ Humans (Ingestion)

Fish *(pathway associated with EFPC fish)*

Fish ÷ Humans (Ingestion)

Population Size

Available data suggest there were approximately ten farms in the EFPC floodplain over the history of ORR operations. Based on this number of farms, the number of farm family individuals was estimated to range between 10 and 50 during a given year from 1950 and 1990. Assuming that in any given year, 1/15 of the population left and was replaced by new residents (Israeli and Nelson 1992), the total population size between 1950 and 1990 was estimated to be between 40 and 200 individuals. It is not known how many people may have been exposed to mercury in beef from cattle that were grazed in the floodplain and sold for slaughter.

6.2.1.5 Oak Ridge Community Near-Floodplain Residents

Activities and Characteristics of the Population

Two "near-floodplain resident" locations were selected as representative of exposures to individuals (adults and children) who lived in residential areas outside of the floodplain yet near EFPC, who may have been exposed to mercury volatilized from EFPC. In addition to direct inhalation of airborne mercury, it is assumed that these residents may have had their own backyard gardens and consumed fruits and vegetables containing mercury taken up from air. The "near-floodplain resident" population locations evaluated in this assessment are on the north side of Tennessee Highway 95 (the Oak Ridge Turnpike)– Community Population #1 is located on Louisiana Avenue just northeast of the intersection of Louisiana Avenue and Lincoln Road, approximately 200 yards north of the floodplain, and Community Population #2 is located off of Jefferson Avenue, between Robertsville Road and Livingston Road and across from Johnson Road, approximately one-quarter mile from EFPC (Figure 1-2).

In this assessment, it was assumed that near-floodplain residents could have been exposed to mercury through the pathways listed below.

Exposure Pathways Evaluated for Near-Floodplain Residents

Air *(pathways from volatilization from EFPC)*

Air ÷ Humans (Inhalation)
Air ÷ Above-ground Fruits and Vegetables ÷ Humans (Ingestion)

Population Size

The population of the city of Oak Ridge reached a peak of about 77,000 individuals in 1945, then decreased rapidly and stabilized at around 27,000 to 30,000 individuals (Overholt 1987, Broughton 1989). Historical population estimates are presented in Table 6-2.

Table 6-2: Estimated Population of the City of Oak Ridge between 1943 and 1990

Year	Population
1943	0
1945	77,000
1950	30,200
1960	27,200
1970	28,300
1980	27,700
1990	27,310

Sources: Overholt (1987), Broughton (1989)

As originally planned, most of the residential areas of the city of Oak Ridge were north of the Oak Ridge Turnpike (Skidmore Owings & Merrill 1948). In the 1948 Oak Ridge Master Plan, the neighborhoods west of Illinois Avenue and what was then Gamble Road were designated as neighborhoods 7, 8, and 9. In the 1940s, housing in these neighborhoods closest to EFPC consisted primarily of dormitories or other multiple-family apartment-type dwellings. Approximately 20 of these units were located within one-quarter mile of EFPC in this area, on the flat land close to EFPC. Each of these dormitories housed an average of about 130 people (Skidmore Owings & Merrill 1948). However, by the late 1940s, approximately three-fourths of these units were no longer used for housing (Skidmore Owings & Merrill 1948). It is assumed that as many as 600 individuals may have been living in these units in the early 1950s, and that the majority were single male workers. It is assumed that these individuals did not have gardens.

In the early 1950s, about 30 "garden apartment" buildings with a total of 453 apartments were built just south of the Oak Ridge Turnpike, west of the present day location of Illinois Avenue (Skidmore Owings & Merrill 1948, USGS 1952). Apartments of this type were designed to house an average of between two and three people per unit. Based on these figures, it is estimated that about 900 to 1,300 individuals lived in these units in the early 1950s. It is assumed that these individuals did not have gardens, or that they had small container gardens. By 1968, most of the dormitories had been removed, although the apartment buildings remained (USGS 1968a).

Based on housing units shown on USGS quadrangle maps from 1952 and 1953 (Windrock and Bethel Valley Quadrangles, USGS 1953a, b), it is estimated that there were about 250 detached homes within one-quarter mile of EFPC in west Oak Ridge (west of Illinois Avenue) in the early 1950s. However, data in the 1948 Master Plan indicate that about two-thirds to three-fourths of these units were two-family homes, with the remainder being single-family homes. Thus it is estimated that units for about 400 to 450 families existed in this area in the early 1950s. By 1968, approximately 40 to 50 new homes had been built within several hundred yards of EFPC off Lynwood Road, and about 150 new homes were added within

one-quarter mile of EFPC in the Oak Hills Estates area (USGS 1968a). Based on these data, it is estimated that during the 1960s, 1970s, and 1980s, detached homes for about 600 to 650 families existed in this area.

Average household sizes in Oak Ridge have decreased over the years. In 1948, the average household size was 3.34 individuals (Skidmore Owings & Merrill 1948). By 1970, the household size had decreased to 3.00 individuals, and by 1980 it had further decreased to 2.50 individuals (Broughton 1989). Based on these figures, it was estimated that the population size within one-quarter mile of EFPC during a given year between the early 1950s and the early 1990s was between 2,000 and 3,500 individuals. Assuming one-sixth of these individuals moved every year and were replaced by new residents, it is estimated that between 15,000 and 30,000 individuals lived in this area between 1950 and 1990.

6.2.1.6 Downstream Angler Populations

Activities and Characteristics of the Populations

Potential exposure to mercury through fish consumption was evaluated for individuals who fished in Clinch River/ Poplar Creek and Watts Bar Reservoir, in addition to individuals from the Scarboro Community and EFPC Floodplain Farm Family populations who were assumed to occasionally catch and consume fish from EFPC. Studies of mercury in fish downstream from EFPC show that concentrations of mercury in fish were frequently above fish advisory levels (see Table 5-7 and Appendix J). In general, exposure to mercury through fish consumption is recognized as one of the primary pathways of exposure to mercury worldwide (WHO 1976; Clarkson 1990; Fitzgerald and Clarkson 1991; ATSDR 1994).

Historical fish consumption rates for four angling *populations* who fished Clinch River/ Poplar Creek or Watts Bar Reservoir were estimated (see Appendix L). In addition, consumption of fish from these two river systems was evaluated for three *categories* of fish consumers based on the number of fish meals that these individuals consumed (for an adult female, an average fish meal is assumed to be 170 g).

Angling *populations* downstream of EFPC that were evaluated were:

- Watts Bar Reservoir Commercial Anglers,
- Watts Bar Reservoir Recreational Anglers,
- Clinch River/ Poplar Creek Commercial Anglers, and
- Clinch River/ Poplar Creek Recreational Anglers.

The term “recreational angler” is used to describe individuals who caught fish for personal consumption, while “commercial angler” refers to full-time anglers who used commercial fishing gear to catch large amounts of fish for commercial sale. It is assumed that commercial anglers consumed fish they were unable to market. These angling populations are characterized in detail in Appendix L. Historical minimum, mean, and maximum consumption rates estimated angler populations are summarized in Table 6.3.

Table 6-3: Fish Consumption Rate Distributions for Angler Populations[a]

Population	Lower-Bound Consumption ($g\ d^{-1}$)	Arithmetic Mean Consumption ($g\ d^{-1}$)	Upper-Bound Consumption ($g\ d^{-1}$)
Watts Bar Reservoir Commercial Angler	0.97	24	90
Clinch River/ Poplar Creek Commercial Angler	0.090	2.2	8.4
Watts Bar Reservoir Recreational Angler	1.2	30	110
Clinch River/ Poplar Creek Recreational Angler	0.76	18	65
East Fork Poplar Creek Angler	0.047	1.2	4.6

[a] The shapes of the consumption rate distributions for the Commercial and Recreational Angler populations are lognormal; the upper- and lower-bound values shown represent the 5th and 95th percentiles of the distributions.

Categories of fish consumers that were evaluated were as follows:

- *Category 1*: >1 to 2.5 fish meals per week (equivalent to approximately 24 to 61 $g\ d^{-1}$, assuming 170 g per fish meal)
- *Category 2*: > 0.33 to 1 fish meals per week (or more than 1 meal every 3 weeks to 1 meal per week, equivalent to approximately 8.0 to 24 $g\ d^{-1}$, assuming 170 g per fish meal)
- *Category 3*: 0.04 to 0.33 fish meals per week (or 1 meal every six months to 1 meal every 3 weeks, equivalent to approximately 0.97 to 8.0 $g\ d^{-1}$, assuming 170 g per fish meal)

Based on the historical fish consumption rates estimated for angler populations downstream of Y-12 presented in Table 6-3, Category 3 is the only category of fish consumer likely to have existed for EFPC. Interviews with Oak Ridge area residents, including residents of the Scarboro Community, suggest that the maximum rate of consumption of fish from EFPC was about one meal per month (DaMassa 1995).

Population Size

Tennessee Valley reservoirs, including Watts Bar Reservoir, and their tailwaters are productive and popular recreational fisheries. In addition, Watts Bar Reservoir has been commercially fished since its impoundment. For Watts Bar, the project team conservatively assumed that there were a total of seven full-time commercial anglers in a given year (Hargis 1968), and that each year one angler stopped activity and another commenced activity. The resulting estimate of the total commercial angler population potentially exposed between 1945 and 1995 may have been as large as 57 anglers. Because it was

reasonable to assume that members of commercial anglers' families may have consumed fish caught by the angler, an average household size of 3.1 individuals (average of household sizes between 1960 and 1990) was assumed to derive an estimate of 180 individuals who may have consumed all of their fish from this source over the duration of the ORR operations. Given the uncertainties in this estimate, the population size was estimated to be between 100 to 300 persons.

In the absence of recreational angler population data for Watts Bar, a base population size was estimated using data on level of effort (total trips) and the number of trips that the average angler might have taken in a year. TWRA (1993) reported 150,698 trips in 1990 for Watts Bar. Based on the assumption that the average Tennessee angler took 14.6 fishing trips to lakes and reservoirs in a single year (USDOI 1993), it was estimated that approximately 10,321 anglers fished Watts Bar in 1980. Based on local census data for that year, this angler population represented approximately six percent of the total population of Anderson, Loudon, Meigs, Rhea and Roane Counties at that time. To develop population estimates, it was first assumed that, in any given year, approximately six per cent of the total population, reported by the census for these counties, fished Watts Bar. This factor was applied to the total population data for each of the census years during the time period of interest to estimate population size in a given year. Due to changes in residence and fishing behavior over time, however, the same individuals could not be assumed to fish the reservoir during every year. Thus, it was important to include an annual turnover rate when estimating the total number of potentially exposed individuals over the period of interest. By adjusting farm family residence times reported by Israeli and Nelson (1992) to reflect a lower rate of inter-regional mobility, a mean exposure duration of 31 years was derived. It was assumed that in any given year, 1/31 of the angler population was replaced by new anglers. Using this assumption, assuming 3.1 individuals in the typical angler household, and accounting for increases in the local population from year to year, it was estimated that approximately 132,000 individuals may have consumed recreationally-caught fish from Watts Bar between 1945 and 1995. Based on this analysis, a population size ranging from 100,000 to 300,000 persons was estimated for Watts Bar.

However, it is unlikely that the Clinch River/ Poplar Creek system was commercially fished to any great degree due to limited access to larger boats and the proximity of the Watts Bar commercial fishery. While it is possible that the Clinch River/ Poplar Creek system was used for commercial fishing, the catch by full-time commercial anglers was probably small compared to the catch from the larger, more productive, and more accessible Watts Bar Reservoir (Ebert 1996). For the Clinch River /Poplar Creek, the project team used the data reported for Watts Bar combined with data from Todd (1990) that reported that only 20 percent of commercial anglers fished rivers. If this percentage was applied to the seven anglers estimated for Watts Bar Reservoir, the resulting estimate was one commercial angler using Clinch River/Poplar Creek in a given year. It was conservatively assumed that every seven years another angler began to fish the area, resulting in a total commercial angler population size estimated at eight individuals between 1945 and 1995. Assuming 3.1 individuals were in the typical commercial angler household, it was estimated that a total of 25 individuals may have consumed all of their fish from this source on the Clinch River/Poplar Creek during operation of the ORR. Given the uncertainties in this estimate, the project team estimated a population size ranging from 10 to 30 individuals.

There were no data available to provide estimates of the number of anglers who may have used Clinch River/Poplar Creek as a fishery. U.S. Fish and Wildlife (USDOI 1993) data for Tennessee indicated that a total of 479,600 state residents fished large lakes or reservoirs during 1991. In that same year, 338,300 anglers fished the state's rivers or streams. Based on those data, it appeared that he number of anglers who fished rivers and streams was approximately 70 percent of the number of anglers who fished lakes and reservoirs. Applying this percentage to the estimated 132,000 persons consuming recreationally-caught fish from Watts Bar, resulted in an estimated population size for Clinch River/Poplar Creek anglers of 92,000. A population size ranging from 30,000 to 100,000 individuals who consumed fish as a result of recreational angling on Clinch River or Poplar Creek during the years the ORR has been in operation was assumed for this analysis.

6.2.2 Potentially Sensitive Population Subgroups

Typically, in human health risk assessment, exposures are evaluated for two population subgroups– adults and children– because of differences in exposure characteristics and intake rates. Children are often considered to be more susceptible to adverse health effects from exposure to contaminants than adults, in part because of the relatively higher ratio of intake rate to body weight in children as compared to adults. In this assessment, exposures to both adults and children were considered.

As discussed briefly in Section 5.0 and in greater detail in Section 11.0, each of the several forms of mercury associated with environmental exposures has a different health effect endpoint. For example, chronic (long term) exposure to high levels of elemental mercury, such as in workplace exposures, has been associated with central nervous system (CNS) effects in some individuals, including tremor, changes in personality and behavior, loss of memory, and depression (Goyer 1996), while in some individuals, chronic exposure to high concentrations of inorganic (mercuric) mercury has been shown to affect kidney function.

The primary effects of exposure to organic (methyl) mercury are also on the CNS, although studies of methylmercury exposures in several populations suggest that the fetus may be particularly susceptible to adverse health effects from methylmercury exposure. This is because organic mercury can cross the placenta and the blood-brain barrier, so that women pass methylmercury to the fetus during pregnancy, and because of the sensitivity of the developing nervous system (Marsh et al. 1980; Marsh et al. 1987; Clarkson et al. 1985; Kjellstrom et al. 1986; Skerfving 1988; Clarkson 1990). Although investigations of acute and subchronic exposures to high concentrations of methylmercury in Minamata, Japan and Iraq showed an increase in CNS effects in children exposed *in utero*, these reports were associated with no or slight transient symptoms in the mother (Clarkson et al. 1985; Marsh et al. 1987 Clarkson 1990).

There is some evidence that sensitivity of the developing nervous system to methylmercury exposure may continue shortly after birth (i.e., postnatal). A study of methylmercury exposures in juvenile mice suggests that postnatal exposure may result in developmental effects, but no dose-response data are available (i.e., data linking a specific level of exposure to a specific response; Sager et al. 1984; Stern 1993). Observations of developmental effects resulting from postnatal methylmercury exposure have not been confirmed in humans.

Data indicating an increased risk of reproductive effects following ingestion of inorganic mercury have not been identified. Though the database is limited, there is some evidence from workplace exposure and animal studies that the incidence of adverse reproductive effects may increase following exposures to high concentrations of airborne elemental mercury (ATSDR 1997; IRIS 1998). However, other studies have detected no adverse reproductive effects, and available data are not sufficient to evaluate dose-response relationships. Nonetheless, because very young children and the developing fetus may be more sensitive to exposure to mercury than older populations, exposures to these potentially sensitive population subgroups were considered in this assessment.

Childhood exposures were also evaluated for students of Robertsville School, a junior high school located adjacent to the floodplain serving grades 7 through 9. For this population group, exposures were evaluated for children assumed to be between 12 and 15 years of age.

Data from the 1948 Oak Ridge Master Plan indicate that in the late 1940s, approximately 20% of the population of Oak Ridge were women between 20 and 35 years of age, and 17.5% were children under the age of 4. It is assumed that approximately 15% were children between 6 months and 3 years of age.

The population subgroups evaluated for each of the reference population groups are summarized in Table 6-4.

Table 6-4: Population Subgroups Evaluated in the Mercury Dose Reconstruction

Reference Population	Subgroups Evaluated	Approximate Population Size per Year (Approximate Total Population Size) [a]
Wolf Valley Farm Family	Adult Child (6 mo - 3 yrs)	30 - 100 (100 - 350)
Scarboro Community Residents	Adult Child (6 mo - 3 yrs)	800 - 1,200 (6,000 - 10,000)
Robertsville School Children	Child (12 - 15 yrs)	*general students* [b]: 1,500 - 2,000 (20,000 - 30,000) *recreational users of EFPC* [c]: 5 - 20 (100 - 300)
East Fork Poplar Creek Floodplain Farm Family	Adult Child (6 mo - 3 yrs)	10 - 50 (40 - 200)
Oak Ridge Community Residents	Adult Child (6 mo - 3 yrs)	2,000 - 3,000 (15,000 - 30,000)
EFPC Angler	Adult Child (6 mo - 3 yrs) *In utero* exposure[d]	(<100)
Watts Bar Commercial Angler	Adult Child (6 mo - 3 yrs) *In utero* exposure[d]	20 - 30 (100 - 300)
Watts Bar Recreational Angler	Adult Child (6 mo - 3 yrs) *In utero* exposure [d]	10,000 - 30,000 (100,000 - 300,000)
Clinch River/ Poplar Creek Commercial Angler	Adult Child (6 mo - 3 yrs) *In utero* exposure [d]	1 - 5 (10 - 30)
Clinch River/ Poplar Creek Recreational Angler	Adult Child (6 mo - 3 yrs) *In utero* exposure [d]	3,000 - 10,000 (30,000-100,000)
EFPC Anglers	Adult Child (6 mo - 3 yrs) *In utero* exposure [d]	10 - 30 (30 - 100)

a Estimated total population size during a given year. For the Wolf Valley, Scarboro, EFPC floodplain farm family, and Oak Ridge community residents, adult females of child bearing age and children 6 mo to 3-years of age were assumed to comprise approximately 20% and 15% of the total population, respectively.

b Assumed to be exposed to mercury in air and contaminated floodplain soil.

c Assumed to be exposed to mercury in EFPC surface water and sediment.

d The number of fetuses that may have been affected (average doses above the NOAEL) was estimated based on birth rates, the fraction of women of childbearing age, their fish consumption rates, and annual fractions of consumers that exceeded the NOAEL for *in utero* exposure. The estimate is uncertain, but is nearer to 100 than to 1,000.

Estimated doses for each subgroup differ because each was evaluated using exposure characteristics specific to that population (such as body weights, ingestion rates, and inhalation rates). In addition, different exposure times were used to develop average intake rates. For example, chronic exposures to adults and children were evaluated by averaging exposures over one year. Data from methylmercury exposures in Minamata and Iraq suggest that the late embryonic and fetal stages, particularly in the second trimester of pregnancy, may be the periods of greatest vulnerability to mercury exposure during gestation (Cox et al. 1989; Stern 1993).

6.2.2.1 Evaluation of Doses to Adults

Exposures to adults were evaluated by characterizing intake rates by an adult female between 18 and 35 years of age. There is no evidence that adult females are more sensitive to adverse effects from mercury exposure than adult males, although maternal exposure to the fetus may represent a sensitive exposure. Exposures to this population subgroup focused on adult females because:

- Evaluation of doses to adult females allows characterization of possible *in utero* effects.

- Body weight and intake rates vary significantly between males and females in this age range. By focusing on one gender and age range, the uncertainty bounds on exposure parameters used in the Monte Carlo uncertainty analysis of estimated dose are narrowed. However, since doses are normalized to a per-kilogram of body weight basis, it is assumed that estimated doses calculated for an adult from this gender and age group are representative of exposures to all adults.

- It was assumed that during the 1950s and 1960s, adult females in the Oak Ridge area were likely to spend a greater amount of time at home– where they could have been exposed to releases of mercury that migrated off-site from the ORR– than adult males.

Food intake rates and other exposure characteristics typical of adult females from the rural South during the 1950s and 1960s were used, when available, to characterize exposures to this group. Exposures were averaged over a period of one year.

6.2.2.2 Evaluation of Doses to Children

For most of the population groups, exposures to children were evaluated assuming exposure to a male or a female child less than three years of age. Exposures to this population subgroup focused on exposures to young male and female children combined because:

- Limited data from toxicological studies suggest that the CNS of very young, developing children may be potentially sensitive to exposure to mercury.
- The size (e.g., body weight) and intake rates of males and females in this age group are similar; therefore, PDFs describing exposure characteristics of young children were defined for males and females combined.

It was assumed that children of this age were likely to spend the majority of their time in and around the home and were not likely to engage in unsupervised play away from the home (e.g., playing in EFPC). Food intake rates and other exposure characteristics typical of young children from the rural South during the 1950s and 1960s were used when available. Exposures were averaged over a period of one year.

Robertsville School is and has historically been a junior high school; thus, exposures to children attending this school were evaluated assuming exposure to a male child between 12 and 15 years old. Although there is no evidence that males in this age group are more sensitive to adverse effects from mercury exposure than females, exposures to this population subgroup focused on exposures to male children because some exposure characteristics (e.g., inhalation rates) vary significantly between males and females in this age group. By focusing on one gender and age range, the uncertainty bounds on exposure parameters used in the Monte Carlo uncertainty analysis of estimated dose are narrowed.

It was assumed that estimated doses calculated for a child from this gender and age group (presented in mg per kg body weight per day) are representative of exposures to all children in this population group. Exposures to this population group were evaluated assuming exposure to mercury occurred only while the children were at or near the school. It was assumed that children of this age had a greater level of independence than younger children and so potentially spent a greater amount of time in unsupervised recreational activities (e.g., in and near EFPC), and may have spent time at or near the school grounds on weekends or after school. Exposures were averaged over a period of one year.

6.2.2.3 Evaluation of Doses to Unborn Children (*In Utero* Exposure)

In utero exposure of the developing fetus to methylmercury from consumption of contaminated fish was evaluated by characterizing maternal intake rates. In this assessment, maternal intake was evaluated by characterizing exposure to adult females of child bearing age (i.e., between 18 and 35 years of age). Food intake rates and other exposure characteristics typical of adult females from the rural South during the 1950s and 1960s were used when available.

6.3 Parameters and Equations to Estimate Dose

As discussed in Section 2.1, health hazards associated with exposure to a chemical are related to the amount, or dose, of chemical absorbed into the body. Parameters used to estimate the dose of mercury received through exposure pathways identified in Sections 6.1.1 through 6.1.4 are described in detail in Section 7.0 (Estimation of Exposure Point Concentrations), Section 8.0 (Characterization of Transfer of Mercury to Vegetation, Milk and Meat), and Section 9.0 (Identification of Parameter Distributions to Characterize Exposure). In addition, tables in Section 9.0 present the equations used to estimate the dose of mercury through each exposure pathway for each population of interest.

7.0 ESTIMATION OF EXPOSURE POINT CONCENTRATIONS IN SURFACE WATER, AIR, SOIL/SEDIMENT, AND FISH

This section describes how historical exposure point concentrations of mercury were calculated for different environmental media near the ORR. These exposure point concentrations were modeled using historical release data or were based on actual measurements in the off-site environment. This section also describes the assumptions used to characterize the uncertainty and variability in estimated exposure point concentrations, and defines the PDFs used in the dose calculations.

Specifically, this section describes:

- **Exposure point concentrations in surface water,** based on measurements of mercury concentrations in EFPC and modeling of downstream dilution and loss of mercury to other compartments including air and sediment;

- **Exposure point concentrations in air**, based on measured and estimated building air concentrations and ventilation rates and estimated volatilization of mercury from EFPC, and dispersion modeling to locations of off-site reference populations;

- **Exposure point concentrations in EFPC floodplain soil and sediment**, based on measurements of mercury concentrations in surface and core samples; and

- **Exposure point concentrations in fish**, based on measured and estimated concentrations in downstream fish.

Assumptions used to characterize exposure point PDFs for each medium are described in the following sections.

7.1 Mercury Concentrations in Surface Water

During the years of peak mercury releases from Y-12, few measurements were made of mercury concentrations in surface water downstream of Y-12. The K-25 Technical Division (Kwasnoski and Whitson 1955-1961) took spot measurements of mercury concentrations in EFPC surface water just upstream of the confluence of EFPC and Poplar Creek weekly between 1955 and 1961 (Table 7-1). Between 7 and 26 measurements were taken each quarter. Comparisons of these measurements to water samples collected at the Y-12 discharge point during the same time period (described in Section 4.5) indicate that concentrations in EFPC near the EFPC/ Poplar Creek junction ranged from approximately 1% to 60% (average 11%) of concentrations measured directly below the discharge point. This is the only historical sampling program during which water samples were collected routinely in EFPC downstream from Y-12 and analyzed for mercury.

The decrease in mercury concentrations in EFPC near the confluence of EFPC and Poplar Creek, compared to concentrations at Y-12, is likely primarily due to the diluting effect of additional drainage into EFPC. However, some of the mercury released from Y-12 was also "lost" to other environmental compartments such as air or sediment. For example, mercury has a high affinity to bind to sediment. Some of the mercury released from Y-12 adsorbed onto suspended particles and deposited in sediments in EFPC or, during flood events, in the EFPC floodplain (TVA 1985b). During subsequent storms and periods of high water flow, some of this mercury would have become resuspended and been carried downstream to Watts Bar Lake (TVA 1985b). Because of the relatively high vapor pressure of elemental mercury, volatilization of mercury from surface water was also likely to have occurred to some extent.

It is difficult to quantify how much of the mercury originally released from Y-12 was lost to different compartments (e.g., sediment or air). As discussed above, minimal data are available on mercury concentrations in EFPC downstream of the Y-12 Plant. Further, the rate at which mercury transfers between compartments is not well characterized, largely because the fate and transport of mercury in the environment is extremely complex. To gain a general understanding of the relative magnitude of the mercury lost from EFPC, the project team calculated the concentration of mercury expected at the confluence of EFPC and Poplar Creek, assuming no net loss of mercury from surface water and a decrease in concentrations due to dilution only, and compared these concentrations to those measured by Kwasnoski and Whitson (1955-1961) during the same time periods. Based on these calculations, the net loss of mercury to other compartments between Y-12 and the confluence was approximated. The results of these calculations were then used to approximate surface water concentrations at the locations of the downstream populations along EFPC during subsequent years.

Approaches for modeling water concentrations downstream of Y-12, and results of this modeling, are described in the following sections. Details of these calculations are provided in Appendix M.

7.1.1 Approach and Assumptions

Surface water concentrations at downstream locations in EFPC were estimated assuming that concentrations were reduced due to dilution and loss to sediment and air. Specifically, downstream concentrations were calculated by multiplying the annual average concentration at the Y-12 discharge point by a dilution ratio, to account for the increase in the total water volume in EFPC, and by the fraction of the total mercury mass assumed to remain after loss to other compartments:

$$\textit{EFPC conc.}\ (\textit{mg}\ L^{-1}) = \textit{Y-12 conc.}\ (\textit{mg}\ L^{-1}) \times \textit{Dilution ratio} \times (1.0 - \textit{Fraction lost to other compartments}) \qquad (7.1)$$

Table 7-1: Comparison of Mercury Concentrations in Surface Water at the Y-12 Discharge and the EFPC/ Poplar Creek Junction

Date	Average Concentration in Y-12 Effluent (µg/L) [a]	Average Concentration at EFPC/PC Junction (µg/L) [b]	Percent of Release Concentration
Sept. 1955	1,240[b]	252	20%
4th Q, 1955	700	427	61%
1st Q, 1956	360	141	39%
2nd Q, 1956	640	171	27%
3rd Q, 1956	1460	94	6.4%
4th Q, 1956	1010	181	18%
1st Q, 1957	1610	382	24%
2nd Q, 1957	2490	172	6.9%
3rd Q, 1957	3020	113	3.7%
4th Q, 1957	1810	96	5.3%
1st Q, 1958	3650	152	2.2%
2nd Q, 1958	3060	130	4.2%
3rd Q, 1958	1250	54	4.3%
4th Q, 1958	1370	72	5.3%
1st Q, 1959	1020	25	2.5%
2nd Q, 1959	740	8.4	1.1%
3rd Q, 1959	750	8.3	1.1%
4th Q, 1959	200	14	9.5%
1st Q, 1960	190	18	9.5%
2nd Q, 1960	200	16	8.0%
3rd Q, 1960	360	5.9	1.6%
4th Q, 1960	400	3.4	0.85%
1st Q, 1961	300	7.6	2.5%
2nd Q, 1961	100	11	11%
3rd Q, 1961	280	5.2	1.9%
4th Q, 1961	120	15	13%
		Average (SD)	**11% (14%)**

a Source = See Appendix I of this report.
b Source = Kwasnoski and Whitson (1955-61)
SD Standard deviation

Calculation of dilution ratios and loss to other compartments is described below.

Calculation of Dilution Ratios

The diluting effect of additional inflow into EFPC at downstream locations was approximated based on the ratio of the initial discharge volume at Y-12 (in cubic feet per second, cfs) to the estimated water volume in the creek at the downstream location (assumed to be equal to the volume of the initial discharge plus the volume of additional inflows to the creek):

$$Dilution\ ratio = \frac{Y\text{-}12\ discharge\ volume\ (cfs)}{Y\text{-}12\ discharge\ volume\ (cfs) + EFPC\ inflow\ volume\ (cfs)} \qquad (7.2)$$

The inflow volume was estimated from the area of the drainage basin and the rate of precipitation runoff:

$$EFPC\ inflow\ (cfs) = Drainage\ basin\ area\ (mi^2) \times Precip\ (in.) \times Runoff\ (\%) \times \frac{0.07367\ ft^2/mi^2}{s/y} \qquad (7.3)$$

Since 1958, the city of Oak Ridge waste water treatment plant (WWTP) at EFPC Mile 8.3 has augmented flow in EFPC below the treatment plant's discharge into the creek (TVA 1985d). Discharge from the WWTP between 1961 and 1964 ranged from 3 to 10 cfs (USGS 1967). However, the populations evaluated in this assessment that are assumed to have contacted EFPC resided upstream of the WWTP. Therefore, discharges from the WWTP were assumed not to contribute to dilution at these locations.

Data on the size of the drainage basin above several points on EFPC, including upstream of New Hope Pond, at several TVA flow measuring stations, and at a USGS flow measuring station, are presented by TVA in their Instream Contaminant Study (TVA 1985b) (Table 7-2).

Table 7-2: Drainage Areas for EFPC Measuring Stations Used to Estimate Dilution of Mercury in EFPC Surface Water [a]

EFPC River Mile	Drainage Area (mi^2)	Remark
14.7	1.25	New Hope Pond
14.36	1.69	TVA Gage Site
10.0	8.72	TVA Gage Site
6.89	13.9	TVA Gage Site
3.3	19.5	USGS Gage Site
0.03	29.8	TVA Gage Site

a Source = Instream Contaminant Study (TVA 1985d)

Based on these figures, the estimated area of the EFPC drainage basin, downstream of New Hope Pond/ Lake Reality (at EFPC Mile 14.7), is approximately 28.6 mi^2 (29.8 - 1.25 mi^2). Runoff from the drainage area above New Hope Pond is assumed to be included in measurements of the Y-12 discharge volume.

The annual average precipitation in the Oak Ridge area between 1948 and 1964 was 53.9 inches (USGS 1967). During a similar period (1936-1960), the estimated annual average runoff at the USGS EFPC gaging station at EFPC Mile 3.3 was 21.7 inches (USGS 1967), or about 40% of the annual precipitation.

There is some uncertainty about the true value of each of the input parameters to the Equation 7.3. PDFs for these inputs were defined as follows:

- Annual precipitation runoff to other creeks near EFPC, including Emory Valley Creek, Scarboro Creek, Poplar Creek, and Bear Creek, between 1936 and 1960 ranged from 21.7 to 25.2 inches (USGS 1967), or about 37% to 46% of annual precipitation. It was therefore assumed that annual runoff at different reference population locations may have varied from estimates for the USGS EFPC gaging station by ±10%.

- The uncertainty in the area of the drainage basins, based on possible measurement errors and the accuracy of linear extrapolation to locations between the river miles listed above, was assumed to be ±10%.

- During 1935-1959, annual average precipitation rates measured by Union Carbide at K-25 and ORNL were 57.85 inches and 51.52 inches, respectively (USGS 1967), compared to 53.9 inches during approximately the same period in the city of Oak Ridge. It was therefore assumed that annual precipitation at different reference population locations may have varied from measurements in the city of Oak Ridge by ± 5%.

Calculation of Loss to Other Compartments

The mass of mercury lost from EFPC surface water was approximated by determining the "expected" mercury concentration at the EFPC/ Poplar Creek junction, assuming no loss of mercury mass between Y-12 and the junction, and comparing these estimates to concentrations measured near the junction by Kwasnoski and Whitson (1955-1961) during the same time periods. For purposes of these comparisons, "expected" downstream concentrations based on dilution were calculated quarterly, because precipitation runoff in the Oak Ridge area varies significantly depending on the time of year. For example, runoff is much lower during summer months (July to September) than winter months (January to March) (USGS 1967) (Table 7-3).

Table 7-3: Average Quarterly Precipitation and Runoff Measured at Oak Ridge [a]

Quarter	Average Quarterly Precipitation (in/qtr) [b]	Average Quarterly Runoff (in/qtr) [c]	Runoff/ Precipitation (%)
Jan-Mar	17.7	10	56%
Apr-May	11.4	5.1	45%
Jun-Sept	12.3	2.6	21%
Oct-Dec	13.1	4	31%

a Source = USGS (1967)
b Based on data collected at the Oak Ridge U.S. Weather Bureau Station during 1948 - 1964
c Based on data collected at EFPC Mile 3.1 during 1936-1960

Using the estimates of quarterly runoff presented in Table 7-3, the quarterly average water volume near the EFPC/ Poplar Creek junction was approximated quarterly dilution ratios were calculated by dividing the Y-12 release volume by the volume estimated at the junction. "Expected" quarterly concentrations at the junction were then calculated by multiplying the concentration measured at Y-12 by the dilution ratio (Table 7-5). These "expected" concentrations were compared to average concentrations measured at the junction by Kwasnoski and Whitson during the same quarter, to estimate the percent of total mercury released to EFPC that was "lost" to other compartments between Y-12 and the junction (Table 7-5).

Table 7-4: Dilution Ratios Calculated for the EFPC/ Poplar Creek Junction, First Quarter 1956 through Fourth Quarter 1961

Quarter	Y-12 Release Volume (cfs) [a]	WWTP Release Volume (cfs)[b]	Precip. (in.) [c]	Estimated Runoff Volume (cfs) [d]	Estimated Junction Volume (cfs) [e]	Dilution Ratio (at Junction) [f]
1st Q, 1956	18.1	NA	21.5	102.2	120.3	0.15
2nd Q, 1956	17.3	NA	16.4	61.9	79.2	0.22
3rd Q, 1956	16.9	NA	12.9	23	39.8	0.42
4th Q, 1956	18.4	NA	16.3	42	60.4	0.30
1st Q, 1957	20.4	NA	20.8	99	119.5	0.17
2nd Q, 1957	16.2	NA	11.8	44.5	60.7	0.27
3rd Q, 1957	14.7	NA	13.6	24.3	39	0.38
4th Q, 1957	16.7	NA	21.6	55.6	72.3	0.23
1st Q, 1958	14.9	6.5	8.8	41.6	63	0.24
2nd Q, 1958	14.5	6.5	11.7	44.1	65.1	0.22
3rd Q, 1958	12.4	6.5	10.9	19.5	38.3	0.32
4th Q, 1958	12.1	6.5	6.1	15.6	34.1	0.35
1st Q, 1959	13.2	6.5	14.5	68.9	88.6	0.15
2nd Q, 1959	15	6.5	11.4	43.1	64.6	0.23
3rd Q, 1959	14.7	6.5	9.4	16.8	38	0.39
4th Q, 1959	16.2	6.5	15.2	39	61.8	0.26
1st Q, 1960	16.6	6.5	12.6	59.8	82.9	0.20
2nd Q, 1960	15.8	6.5	11.1	41.7	64	0.25
3rd Q, 1960	13.8	6.5	18.9	33.6	53.9	0.26
4th Q, 1960	13.8	6.5	11.8	30.4	50.7	0.27
1st Q, 1961	16.2	6.5	17.1	81.2	104	0.16
2nd Q, 1961	16.7	6.5	15.2	57.1	80.3	0.21
3rd Q, 1961	16.9	6.5	11.5	20.5	43.8	0.38
4th Q, 1961	18.1	6.5	17.5	44.1	68.7	0.26

a Flow measured at the Y-12 release point
b Average flow from the City of Oak Ridge Waste Water Treatment Plant at EFPC Mile 8.3, based on measurements between 1960 and 1964 (USGS 1967)
c Quarterly precipitation measured at the Oak Ridge U.S. Weather Bureau Station (USGS 1967)
d Calculated by multiplying the quarterly precipitation by the average quarterly percent runoff (see Table 7-3) and the area of the EFPC drainage basin (28.55 mi^2) (see Table 7-2).
e Equals the sum of Y-12 release volume + the waste water treatment plant discharge volume + the calculated runoff volume.
f Equal to the Y-12 release volume divided by the estimated junction volume
NA Not applicable

Table 7-5: Calculation of "Expected" Mercury Concentrations near the EFPC/ Poplar Creek Junction and Approximation of Mercury Loss

Quarter	Dilution Ratio (at Junction) [a]	Measured Conc. at Y-12 ($\mu g\ L^{-1}$) [b]	"Expected" Conc. at Junction ($\mu g\ L^{-1}$) [c]	Measured Conc. at Junction ($\mu g\ L^{-1}$) [d]	Percent of Mercury "Lost"[e]
1st Q, 1956	0.15	360	54	141	+160%
2nd Q, 1956	0.22	640	140	171	+22%
3rd Q, 1956	0.42	1460	618	94	-85%
4th Q, 1956	0.3	956	291	181	-39%
1st Q, 1957	0.17	1610	275	382	+39%
2nd Q, 1957	0.27	2420	648	172	-73%
3rd Q, 1957	0.38	3020	1138	ND	ND
4th Q, 1957	0.23	1810	418	96	-77%
1st Q, 1958	0.24	3650	861	152	-82%
2nd Q, 1958	0.22	3060	683	130	-81%
3rd Q, 1958	0.32	1250	404	54	-87%
4th Q, 1958	0.35	1370	484	72	-85%
1st Q, 1959	0.15	1020	151	25	-84%
2nd Q, 1959	0.23	740	172	8.4	-95%
3rd Q, 1959	0.39	750	290	8.3	-97%
4th Q, 1959	0.26	200	53	14	-73%
1st Q, 1960	0.20	190	38	18	-53%
2nd Q, 1960	0.25	200	49	16	-68%
3rd Q, 1960	0.26	360	92	5.9	-94%
4th Q, 1960	0.27	220	60	3.4	-94%
1st Q, 1961	0.16	300	47	7.6	-84%
2nd Q, 1961	0.21	100	21	11	-47%
3rd Q, 1961	0.38	280	108	5.2	-95%
4th Q, 1961	0.26	120	32	15	-53%
Average					**-58%**

ND No data available for this quarter.
a See Table 7-4
b See Appendix I of this report
c Equal to the dilution ratio multiplied by the concentration measured at Y-12
d Average concentration measured at the EFPC/ Poplar Creek junction by Kwasnoski and Whitson (1956-1961)
e Equal to the measured concentration minus the "expected" concentration, divided by the "expected" concentration.

Based on the above calculations, the average quarterly loss of mercury was 58%. However, during the first two quarters of 1956 and the first quarter of 1957, the "expected" concentration was *less* than the concentration measured by Kwasnoski and Whitson. This is counterintuitive, since it is likely that some of the mercury released was lost to other compartments. Averaging loss estimates for the second quarter of 1957 and later results in an average percent loss of 79%.

Clearly, there are a number of uncertainties inherent in this approach, including analytical uncertainties in mercury concentrations measured at Y-12 and the junction, uncertainties in flow measurements, uncertainties about the size of the drainage basin and the fraction of precipitation that runs off into EFPC, and uncertainties about the amount of time required for mercury released from Y-12 to travel to the junction. Further, little is known about the sampling methods used to collect water samples at the EFPC/Poplar Creek junction. For purposes of this evaluation, it was assumed that the analytical methods used to measure mercury in water near the EFPC/ Poplar Creek junction were similar to those used to measure mercury concentrations in water at Y-12 and, therefore, that the precision and accuracy of the analytical methods were the same. However, while there are significant uncertainties in these methods, available data suggest that a significant fraction of the mercury released to EFPC from Y-12 was "lost" to other compartments between Y-12 and the environment. These compartments may have included volatilization to air or deposition to sediment.

Mercury, like a number of other trace metals, tends to bind readily to suspended particulates, such that both bottom and suspended sediments contain significantly higher concentrations than is found in the dissolved phase (Horowitz 1991). Researchers at other sites have indicated that, typically, >99% of mercury transport in surface water systems is associated with the solid phase (Horowitz 1991). As part of the Instream Contaminant Study conducted in 1984, the TVA estimated the total mass of mercury in sediments in EFPC and the EFPC floodplain based on concentrations of mercury measured in transects across the floodplain, at depths ranging from the surface to the boundary between contaminated and uncontaminated sediment. The estimated total mass of mercury in sediments in 1984 was 157,000 pounds. This mass is approximately 57% of the estimated 275,000 pounds of mercury released from Y-12 between 1953 and mid-1984. This value is approximately the same as the average mercury loss estimated above, and approximately 20 to 30% less than the upper-bound estimate. These analyses suggest that a large fraction of the mass of mercury released from Y-12 was lost to other compartments between the Y-12 discharge and the EFPC/Poplar Creek junction, and that most of the mercury that was lost was bound to sediments with only a small fraction of the mercury in the creek likely lost to air.

For purposes of estimating downstream concentrations of mercury in EFPC surface water, it was assumed that approximately 70% of the mercury released from Y-12 to EFPC was lost to other compartments between Y-12 and the junction. The loss of mercury along the length of the creek was assumed to be linear with downstream distance. For purposes of the uncertainty analysis, uncertainty in the percent of mercury mass released from Y-12 that was lost to other compartments was assumed to be ± 30%.

The loss of mercury to other compartments at the location of a given reference population was calculated as follows:

$$\textit{Fraction lost to other compartments} = 0.70 \times \frac{\textit{Reference population downstream distance from Y-12 (mi)}}{14.7\ \textit{mi}} \qquad (7.4)$$

Surface water concentrations calculated for each of the downstream reference population locations on EFPC are described below.

7.1.2 EFPC Surface Water Concentration– Scarboro Community

Annual average surface water concentrations at an EFPC location nearest the Scarboro community (approximately EFPC Mile 14) were calculated assuming the following:

- The area of the EFPC drainage basin was assumed to decrease linearly between EFPC Mile 10.0 and 14.36, resulting in an estimated drainage basin area at Mile 14 of approximately 2.27 mi^2. Subtracting the drainage basin area above New Hope Pond (1.25 mi^2) gives a drainage basin area contributing to inflow from runoff at Mile 14 of 1.02 mi^2.

- The annual average runoff was assumed to be 40% of the annual precipitation, based on measurements at EFPC Mile 3.3.

- The annual average fraction lost to other compartments was assumed to be 0.033 [i.e., 0.70 × (14.7 mi ! 14 mi)/14.7 mi].

Dilution ratios calculated for 1950 to 1990 are presented in Table 7-6. The average dilution ratio for this period was 0.90. Multiplying this ratio by 0.967 (i.e., 1.0 ! 0.033, to reflect loss to other compartments) produces an estimated average downstream water concentration ratio of 0.87 (i.e., for a given year, the average mercury concentration in surface water at EFPC Mile 14 was assumed to be 87% of the average concentration in the Y-12 discharge).

7.1.3 EFPC Surface Water Concentration– Robertsville School

Annual average surface water concentrations at Robertsville School (approximately EFPC Mile 12) were calculated assuming the following:

Table 7-6: Calculated Dilution Ratios at Locations of Downstream Populations

Year	Concentration at Y-12 (mg/L)	EFPC Mile 14 Dilution Ratio	EFPC Mile 12 Dilution Ratio	EFPC Mile 10 Dilution Ratio
1950	0.008	0.90	0.69	0.55
1951	0.016	0.90	0.69	0.56
1952	0.078	0.93	0.77	0.66
1953	0.35	0.92	0.74	0.62
1954	0.22	0.90	0.69	0.56
1955	1.06	0.91	0.71	0.58
1956	0.85	0.90	0.68	0.54
1957	2.22	0.89	0.67	0.53
1958	2.33	0.92	0.74	0.62
1959	0.68	0.91	0.70	0.57
1960	0.24	0.90	0.69	0.55
1961	0.2	0.90	0.69	0.56
1962	0.12	0.91	0.72	0.59
1963	0.086	0.93	0.77	0.65
1964	0.044	0.90	0.68	0.55
1965	0.095	0.90	0.69	0.56
1966	0.043	0.92	0.73	0.61
1967	0.031	0.88	0.63	0.49
1968	0.005	0.93	0.76	0.65
1969	0.006	0.91	0.70	0.57
1970	0.026	0.90	0.68	0.54
1971	0.006	0.90	0.68	0.55
1972	0.001	0.86	0.60	0.46
1973	0.065	0.85	0.59	0.45
1974	0.015	0.85	0.57	0.43
1975	0.001	0.85	0.58	0.44
1976	0.001	0.89	0.65	0.52
1977	0.002	0.88	0.63	0.49
1978	0.001	0.87	0.61	0.47
1979	0.002	0.85	0.58	0.44
1980	0.002	0.92	0.73	0.60
1981	0.002	0.90	0.68	0.55
1982	0.003	0.89	0.65	0.51
1983	0.002	0.91	0.70	0.57
1984	0.0016	0.89	0.67	0.53
1985	NA	0.91	0.72	0.59
1986	NA	0.93	0.75	0.63
1987	NA	0.91	0.72	0.59
1988	0.0019	0.87	0.63	0.49
1989	0.0017	0.85	0.58	0.44
1990	0.0017	0.89	0.67	0.54
AVERAGE		**0.90**	**0.68**	**0.55**

- The area of the EFPC drainage basin was assumed to decrease linearly between EFPC Mile 10.0 and 14.36, resulting in an estimated drainage basin area at Mile 12 of approximately 5.5 mi^2. Subtracting the drainage basin area above New Hope Pond (1.25 mi^2) gives a drainage basin area contributing to inflow from runoff at Mile 12 of 4.25 mi^2.

- The annual average runoff was assumed to be 40% of the annual precipitation, based on measurements at EFPC Mile 3.3.

- The annual average fraction lost to other compartments was assumed to be 0.13 (i.e., 0.70 × 2.7 mi/14.7 mi).

Dilution ratios calculated for 1950 to 1990 are presented in Table 7-6. The average dilution ratio for this period was 0.68. Multiplying this ratio by 0.87 (i.e., 1.0 ! 0.13, to reflect loss to other compartments) produces an estimated average downstream water concentration ratio of 0.59 (i.e., for a given year, the average mercury concentration in surface water at EFPC Mile 12 was assumed to be 59% of the average concentration in the Y-12 discharge).

7.1.4 EFPC Surface Water Concentration– EFPC Floodplain Farm Family

Annual average surface water concentrations at the location of the EFPC Floodplain Farm Family population (approximately EFPC Mile 10) were calculated assuming the following:

- The area of the EFPC drainage basin above EFPC Mile 10 was assumed to be 8.72 mi^2 (TVA 1985b). Subtracting the drainage basin area above New Hope Pond (1.25 mi^2) gives a drainage basin area contributing to inflow from runoff at Mile 12 of 7.47 mi^2.

- The annual average runoff was assumed to be 40% of the annual precipitation, based on measurements at EFPC Mile 3.3.

- The annual average loss to other compartments was assumed to be 0.22 (i.e., 0.70 × 4.7 mi/14.7 mi).

Dilution ratios calculated for 1950 to 1990 are presented in Table 7-6. The average dilution ratio for this period was 0.55. Multiplying this ratio by 0.78 (i.e., 1.0 ! 0.22, to reflect loss to other compartments) produces an estimated average downstream water concentration ratio of 0.43 (i.e., for a given year, the average mercury concentration in surface water at EFPC Mile 12 was assumed to be 43% of the average concentration in the Y-12 discharge).

7.1.5 Characterization of Uncertainty in Surface Water Concentration Estimates

PDFs used in the dose calculations to describe annual average mercury concentrations in surface water at different population locations were defined as follows:

$$C_{water} = C_{water\ (Y\&12\ discharge)} \times Dilution\ ratio\ (\%) \times (100\% - Loss\ to\ other\ compartments) \times C_{water(Unc)} \quad (7.5)$$

Where:

C_{water}	=	Location-specific annual average surface water concentration, used in Monte Carlo uncertainty analysis (mg L^{-1})
$C_{water(Y\text{-}12\ discharge)}$	=	Annual average mercury concentration in Y-12 discharge to EFPC (mg L^{-1})
Dilution ratio	=	Calculated annual average dilution factor at downstream population locations due to contribution of runoff to Y-12 flow rate (unitless)
Loss to other compartments	=	Calculated percentage of mercury mass "lost" to air or sediment between the Y-12 discharge and the population location (unitless)
$C_{water(Unc)}$	=	Uncertainty in calculated downstream water concentration (unitless)

PDFs describing annual exposure point concentrations in surface water were defined based on uncertainties in concentrations and volumes of Y-12 discharge to EFPC, as well as assumptions to reflect uncertainties in the size of the drainage basin, average precipitation and runoff rates, discharge volumes from the Oak Ridge waste water treatment plant, and loss of mercury mass to other compartments, as described above. Data on the precision of the analytical methods for measuring mercury concentrations in discharges to EFPC indicate uncertainties in measured concentrations range from ± 50% in 1953 to ± 10% in 1993, and data on the quality of the flow rate data, as determined by the USGS, indicate that uncertainties in measured annual average discharge volumes from Y-12 ranged from ± 15% in 1953 to ± 10% in 1993. Assumptions used to derive the uncertainty bounds for each of these parameters are described in Appendix M and summarized in Table 7-7.

Table 7-7: Uncertainty Bounds for Characterization of Surface Water Concentrations Downstream from Y-12

Year	Y-12 Conc.	Y-12 Flow Rate	Annual Average Precip.	Annual Average Runoff	Drainage Basin Area	WWT Plant Flow Rate	Loss to other Compart-ments
1953-56	±50%	±15%	±10%	±10%	±10%	NA	±30%
1957-59	±15%	±15%	±10%	±10%	±10%	3-10 cfs (uniform, 1958-59 only)	±30%
1960-61	±30%	±15%	±10%	±10%	±10%	3-10 cfs (uniform)	±30%
1962-67	±40%	±15%	±10%	±10%	±10%	3-10 cfs (uniform)	±30%
1968-82	±20%	±10%	±10%	±10%	±10%	3-10 cfs (uniform)	±30%
1983-93	±10%	±10%	±10%	±10%	±10%	3-10 cfs (uniform)	±30%

NA = Not applicable

7.2 Mercury Concentrations in Air Due to Direct Emissions from Y-12

Measurements of mercury concentrations in air at off-site locations surrounding the ORR are not available for the years of peak mercury releases from Y-12 (1953 - 1962). The earliest available ambient air data for the area surrounding Oak Ridge are from 1986 (Table 5-1; summary data are tabulated in Appendix J). Because concentrations of mercury in air at off-site locations for earlier years are not available, the project team estimated off-site air concentrations for these years using the Industrial Source Complex Short Term (ISCST) air dispersion model with emissions data and local meteorological data (dispersion modeling quantitatively relates contaminant emission concentrations to resulting airborne concentrations at points of interest). Two sources of air emissions were considered for mercury: direct releases from Y-12 and volatilization of mercury from EFPC surface water. The approach used to estimate airborne mercury concentrations at points of interest due to direct releases from Y-12 is summarized below. The approach used to estimate concentrations of mercury in air due to volatilization from EFPC is described in Section 7.3.

Direct air emissions from Y-12 during 1953 to 1962 were modeled to estimate mercury concentrations in air at the location of a residential community in the Wolf Valley area. Wolf Valley is located "down valley" from Y-12, about five miles northeast of Y-12 on the opposite side of the Clinch River. Mercury concentrations in air due to dispersion of direct emissions from Y-12 were also modeled for the Scarboro Community. As described in Sections 6.2.1.1 and 6.2.1.2, Pine Ridge restricts the exchange of air between Y-12 and the Scarboro Community (U.S. Weather Bureau 1953; Gifford 1995). However,

meteorological studies and environmental monitoring programs for airborne contaminants indicate that the ridges in the Oak Ridge area are not perfect barriers to transport of airborne effluents into adjacent valleys. For example, although airborne mercury concentrations have not been measured in the Scarboro Community, concentrations of airborne uranium in Scarboro have been recorded; it is assumed that the uranium originated from Y-12. In this assessment, historical airborne mercury concentrations in the Scarboro Community due to Y-12 releases were estimated using "empirical P/Q's" developed by the Oak Ridge Dose Reconstruction Task 6 team to characterize the relationship between concentrations of airborne uranium in Scarboro and uranium releases from Y-12 (Task 6 addresses releases of uranium from Y-12).

The approach used to estimate off-site ambient air concentrations of mercury due to direct releases from Y-12 and the results of the modeling are described in the following sections.

7.2.1 Estimation of Downvalley Air Concentrations

Emissions of mercury to air from Y-12 were estimated from measurements of mercury in building air and measured and estimated building ventilation rates. The approach and assumptions used and the modeling results are described below.

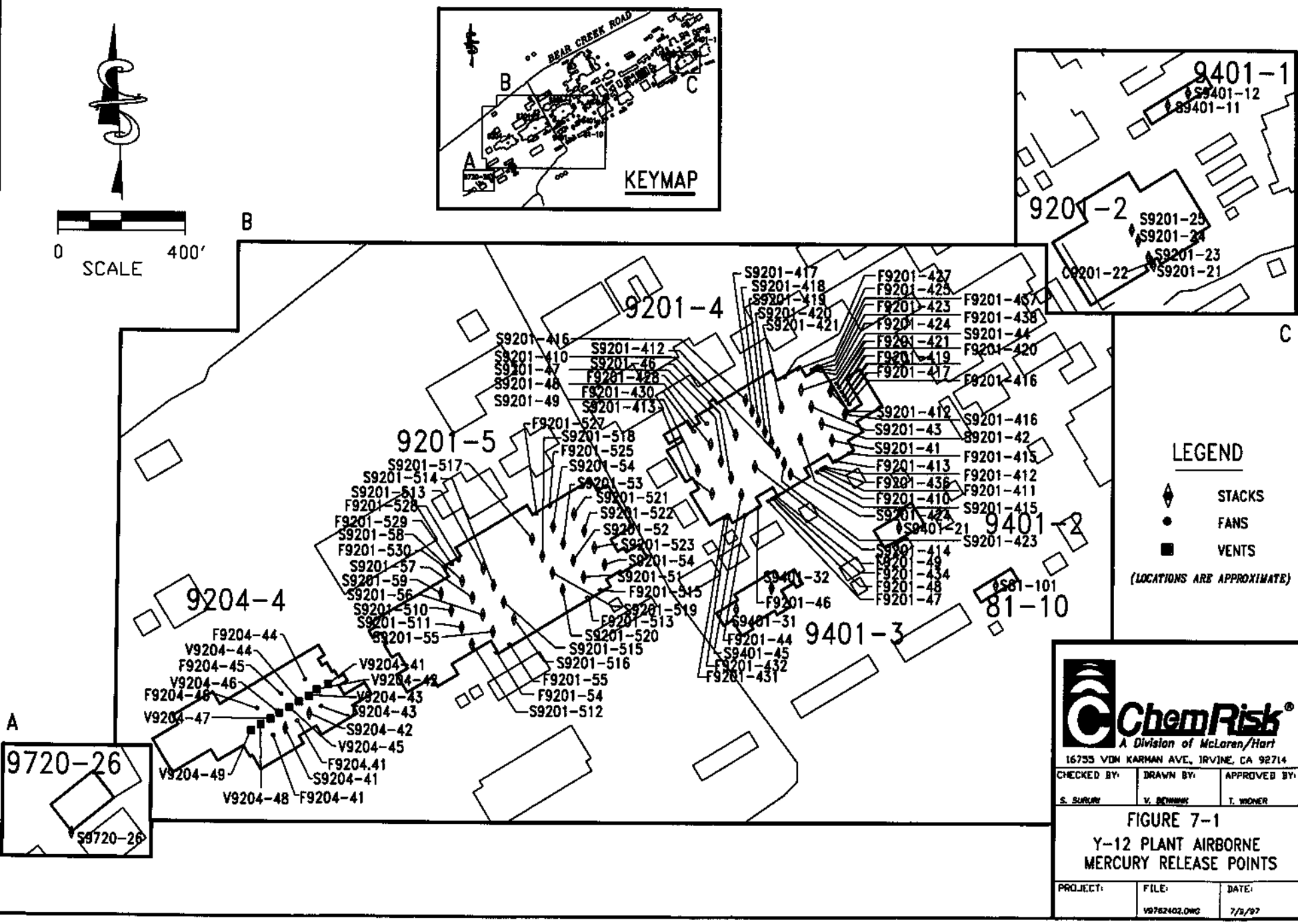
7.2.1.1 Approach and Assumptions

Locations of mercury releases to air from Y-12 modeled in this assessment included Buildings 9201-2, 9201-4, 9201-5, 9204-4, 9401-1, 9401-2, 9401-3, 81-10 and an incinerator west of Building 9720-26. Individual emission sources included 62 stacks, 43 fans, and 9 vent sources**S** these were modeled as point sources. A former Y-12 ventilation engineer determined the locations and volume flow rates (in cubic feet per minute) of individual emission sources using ORR engineering drawings (Choat 1996; see Appendix G). Source locations are shown in Figure 7-1.

Mercury emission rates (in lbs yr^{-1}) were estimated for each source as described in Section 4.4.2. Estimated emission rates are tabulated in Appendix H.

Model Description

Air dispersion modeling was conducted to estimate ground-level exposure point concentrations of airborne mercury at several discrete receptor points, including Wolf Valley, on-site ambient air monitoring stations, and the locations of trees that were analyzed for mercury in tree rings (see discussion, Section 7.2 and Appendix O). The computer model used to simulate air dispersion of mercury emissions was the most recent version of the USEPA-approved Industrial Source Complex Short Term (ISCST3) model, Version 96113 (USEPA 1995c). ISCST3 is accepted by the USEPA as an appropriate air dispersion model for use in relatively flat terrain. The ISCST3 air dispersion model was run to determine annual average airborne mercury concentrations at the receptor points based on a unit emission rate (1 g s^{-1}) from each source. The contribution of each source to the annual average air concentration at each receptor was determined by multiplying the air concentration at the receptor associated with a unit emission by the annual average emission rate (Q) for the source:



ChemRisk®
A Division of McLaren/Hart
16755 VON KARMAN AVE, IRVINE, CA 92714

CHECKED BY:	DRAWN BY:	APPROVED BY:
S. SURUM	V. BOHMAN	T. WIDNER

FIGURE 7-1
Y-12 PLANT AIRBORNE MERCURY RELEASE POINTS

PROJECT:	FILE:	DATE:
	V9762402.DWG	7/8/97

$$C_{ij,n} = C_{1,ij} \times Q_{i,n} \quad (7.6)$$

where:

$C_{ij,n}$	=	Concentration at receptor *j* due to emission from source *i* during year *n* ($\mu g\ m^{-3}$)/(1 g s^{-1})
$C_{1,ij}$	=	Concentration at receptor *j* due to unit emission (1 g s^{-1}) from source *i* ($\mu g\ m^{-3}$)/(1 g s^{-1})
$Q_{i,n}$	=	Mercury emission rate from source *i* for year *n* (g s^{-1})
i	=	Source number
j	=	Receptor number
n	=	Year of emission

The total annual average air concentration at each receptor was then calculated by summing the contributions from all sources:

$$C_{j,n} = \sum_{i=1}^{m} C_{i,j} \quad (7.7)$$

where *m* is the total number of sources.

Model Input Parameters

Inputs to the air dispersion model included:

- **Source parameters** including source location, stack height, release direction, velocity, and exit gas temperature. Source parameters input to the model are presented in Appendix N in Table N-2.

- **Source emission rates in grams per second**. Source emission rates input to the model are presented in Appendix N in Table N-3. Source emission rates were based on a comprehensive review of historical documents, operations records, and interviews with plant personnel, as described in Section 4.

- **Hourly meteorological data** describing wind speed, wind direction, temperature, stability class, and mixing height. During the years of greatest air emissions of mercury (i.e., 1950-1963), hourly meteorological data for the EFPC floodplain are not available. Monthly average data from the Oak Ridge town center station (Station 886) were compared to hourly average data collected from 1987-1992

at the Y-12 East Meteorological station. Based on this comparison, meteorological data from the Y-12 East station for the year 1987 were used to provide hourly wind speed, wind direction, temperature, stability class, and mixing height information to model releases from EFPC.

- **Receptor locations**, including the off-site Wolf Valley residential residence, on-site air monitoring locations, and on-site trees analyzed for mercury concentrations in tree rings.

7.2.1.2 Simulation Results and Uncertainty in Exposure Estimates

Estimated airborne mercury concentrations at the downvalley reference populations and the on-site ambient air samplers due to air releases from Y-12 are summarized in Table 7-8 for the years of peak releases (i.e., 1953-1962). These concentrations are assumed to represent annual average air concentrations.

Table 7-8: Estimated Air Concentrations due to Emissions of Mercury from Y-12, Modeled Using ISCST3 (µg m^{-3})

Location of Interest	1953	1954	1955	1956	1957	1958	1959	1960	1961	1962
Wolf Valley Resident	0.0008	0.0020	0.014	0.0084	0.0037	0.0057	0.0048	0.0022	0.0016	0.0016
Ambient Air Monitoring Station No. 2	0.0080	0.018	0.12	0.077	0.035	0.048	0.040	0.022	0.016	0.016
Ambient Air Monitoring Station No. 8	0.059	0.15	0.71	0.42	0.19	0.27	0.21	0.12	0.090	0.089

PDFs describing air concentrations at the locations of off-site reference populations were defined by bounding annual average air concentrations modeled to each location for a given year ($C_{j,n}$) using information on uncertainties in model inputs (i.e., emission rates) and uncertainties induced by the modeling approach, as follows:

$$C_{air\&Y\&12} \text{ ' } C_{j,n} \times Q_{Unc} \times M_{Unc} \qquad (7.8)$$

where:

$C_{air\text{-}Y\text{-}12}$ = Receptor-specific annual average air concentration due to dispersion of emissions from Y-12, used in Monte Carlo uncertainty analysis (µg m^{-3})

$C_{j,n}$ = Modeled average air concentration at receptor *j* during year *n* due to emissions from Y-12 (µg m^{-3}) (see Table 7-8)

Q_{unc} = Uncertainty in emission rate (equivalent to aggregated uncertainty in measured building air concentration + uncertainty in building ventilation rate)

M_{unc} = Uncertainty in air dispersion model

Assumptions used to characterize emission rate uncertainty and model uncertainty are described below.

Emission Rate Uncertainty

As described above, mercury emissions to air from 114 sources at Y-12 were used in the ISCST3 model to predict air concentrations at the points of interest. For several sources (e.g., Buildings 9201-5, 9201-4, 9204-4, and 9201-2) emission rates were calculated as the product of measured building air concentrations and air flow rates from the building (Appendix N):

$$Q_{i,n}(g\ s^{-1}) = C_{air\&Bldg}\ (mg\ m^{-3}) \times Flow\ rate\ (ft^3\ \min^{-1}) \times 0.0000283\ (m^3\ ft^{-3})(g\ mg^{-1})(\min\ s^{-1}) \qquad (7.9)$$

Uncertainties in emission rates were assumed to result from uncertainties in both measured air concentrations and flow rates. Assumptions used to characterize the uncertainties are described in Section 4.4 and summarized below:

- Uncertainty in mercury air concentrations from Buildings 9201-5, 9201-4, and 9204-4, measured using Y-12 portable mercury vapor detectors, is estimated to be ± 40% (Prestbo 1996).

- Uncertainty in building air flow rates for Buildings 9201-5, 9201-4, and 9204-4 is estimated to be ± 3%, based on ventilation design drawings and assumptions about minor variations in as-built conditions compared to design conditions and minor alterations for spot ventilation in problem areas (Choat 1996). Since there were incomplete drawings of the 9201-2 building ventilation system, uncertainty in the exhaust air flow rate for Building 9201-2 is estimated to be ± 50% (Choat 1996).

- Data on mercury released from several sources (e.g., Buildings 81-10 and the steam plants) were reported as pounds of mercury released per year. The uncertainty about the true mass of mercury released was assumed to be ± 50%.

Aggregating the uncertainties in emission rates as applied to the Wolf Valley exposure population resulted in the assumption that, for a given year, the true annual average concentration at this receptor ranged from ±44% of the predicted concentration.

Model Uncertainty

Uncertainties in model predictions arise from incomplete knowledge or oversimplification of the processes modeled. Little and Miller (1979) reviewed Gaussian plume dispersion models such as ISCST and estimated that, for a highly instrumented flat-field site, ground-level centerline concentration predictions within 10 kilometers of a continuous point-source are accurate within a factor of 1.2. Predictions of annual average concentrations for a specific point on flat terrain and within 10 kilometers of the release point are accurate within a factor of 2. Miller and Hively's more recent review (1987) of Gaussian plume model estimates of airborne radionuclide exposures reiterates the earlier estimates.

The meteorologic data used in the ISCST model in the current assessment are not derived from a highly instrumented site. However, the studies described above suggest that uncertainties in the model predictions range from a factor of 1.2 to 2. The PDF used to characterize ISCST model uncertainty is represented by a lognormal distribution having a geometric mean of 1 and a geometric standard deviation of 1.4 (i.e., 95% of the distribution is within a factor of 2 of the geometric mean).

PDFs used in the calculations to characterize off-site air concentrations at downvalley locations due to dispersion of mercury emissions from Y-12 are summarized in Table 7-9.

Table 7-9: Probability Density Functions for Characterization of Air Concentrations Due to Dispersion of Emissions from Y-12

Parameter	PDF	
	Distribution Type	**Description**
Modeled air concentration ($C_{air\text{-}Y\text{-}12\ (mod)}$)	Point	See model output summary (see Table 7-6)
Model uncertainty (M_{Unc})	Lognormal	Geometric mean = 1 Geometric SD = 1.4
Building air concentration uncertainty ($C_{air\text{-}bldg(Unc)}$)	Uniform	Lower-bound = -40% Upper-bound = +40%
Building ventilation rate uncertainty ($Flow_{Unc}$)	Uniform	Lower-bound = -3% Upper-bound = +3%

7.2.2 Estimation of Scarboro Air Concentrations

No measurements of mercury concentrations in air have been made in the Scarboro Community. However, measurements of airborne uranium at Scarboro, presumably released from Y-12, have been made in recent years, suggesting that some fraction of the airborne releases from Y-12 are transported over Pine Ridge.

Because of the unique characteristics of the topography surrounding the Y-12 facility, a classical air dispersion modeling approach would over-estimate air concentrations at the Scarboro Community 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 the Scarboro Community 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. Mercury concentrations in air at the Scarboro Community, due to direct airborne releases from Y-12, were therefore estimated using an empirical approach based on the ratio between measurements of airborne uranium in the Scarboro area and estimates of uranium releases from Y-12 developed by the Oak Ridge Dose Reconstruction Task 6 team. It is assumed that the relationship between mercury concentrations in air at Scarboro and mercury release rates from Y-12 is the same as the relationship between uranium air concentrations and release rates.

Empirical χ/Q values used to estimate airborne mercury concentrations at Scarboro, based on releases from Y-12, were developed as follows (Equation 7.10):

$$\textit{Empirical } \chi/Q \ (s\ m^{-3}) = \frac{\textit{Uranium Air Concentration Measured at Scarboro } (pCi\ m^{-3})}{\textit{Uranium Release Rate } (pCi\ s^{-1})} \qquad (7.10)$$

Empirical χ/Q's were calculated for calendar years 1986 through 1995 (the years of uranium sampler operation at Scarboro) for two uranium istopes**S** $^{234/235}$U and ^{238}U. Statistical analyses of the annual χ/Q values yields the summary statistics presented in Table 7-10. Tests for lognormality of the data set were inconclusive; consequently, for purposes of estimating mercury concentrations at Scarboro due to direct releases from Y-12, a custom distribution was established using each of the 20 discrete χ/Q values.

Table 7-10: Statistical Analysis of Empirical P/Q Values for Y-12 Uranium Releases

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

For the Task 2 evaluation of mercury concentrations at Scarboro, the empirical P/Q relationship for Y-12 uranium releases was selected as the best alternative for estimating annual average mercury concentrations at that community. Given the complexity of the airborne effluent transport patterns between Y-12 and Scarboro, this measurement-based approach was favored over the mathematical modeling that would be possible within the schedule and budget of this project. The evaluation of the applicability of the empirical P/Q approach included consideration of the fact that, while uranium and mercury were both released from the Y-12 Plant, there were differences in the contaminants and how they were released.

Some key lithium enrichment operations were conducted in buildings that had earlier housed electromagnetic enrichment operations for uranium (e.g., Buildings 9201-2, 9201-4, 9201-5, and 9204-4, all located in the west-central area of the main groupings of Y-12 buildings). Approximately 93% of estimated Y-12 airborne mercury releases occurred from three buildings that had earlier housed key uranium enrichment operations. Uranium forming and machining operations were active after 1952, when Y-12 uranium releases increased significantly before decreasing significantly again in the mid-1960s. While these operations used two of the buildings that had been used for lithium enrichment, some buildings in the north-central area of the site were also used. Both uranium and mercury were released from a wide number of release points on the Y-12 site, rather than from single point sources. The fact that emission sources for both contaminants were distributed over a significant portion of the Y-12 site indicates that variations in the identity of specific sources over time should not strongly influence relative concentrations at the Scarboro Community.

Mercury is considered to have been released to the air from Y-12 operations in the form of elemental mercury vapor. Uranium was released from Y-12 operations in the form of solid particles, most likely in forms such as uranium metal, oxides, or salts. Most airborne mercury releases appear to have been associated with ventilation of buildings that housed lithium enrichment operations or support functions, rather than from process venting or exhausts. Evidence of this includes the huge fans that were added to the walls of Building 9201-5 to reduce concentrations in operating areas. Uranium releases were more

likely to have been discharged via stacks or elevated release points than were mercury releases. At the same time, the "stacks" that were used for uranium emissions were generally not of the tall, 200-foot type like several built at X-10. Rather, they were often roof-top or relatively short stacks that would in most cases be within the zone of building wake effects and would fail to qualify for elevated release point status under regulatory guidance.

In general, increasing the elevation of a release point results in lower concentrations at close-in distances. However, at the same time, increased elevation may have increased the fraction of the releases that carried over Pine Ridge to reach Scarboro. In addition, uranium releases likely experienced greater rates of wet and dry deposition than mercury vapors, due to their particulate nature. So, while uranium releases might have been released at slightly greater elevations than mercury releases, the difference was not likely large enough to have had a significant impact on relative concentrations at Scarboro.

Based on the assessment of uranium transport to Scarboro, a relative concentration factor was applied to annual Y-12 airborne mercury emissions to estimate mercury concentrations at Scarboro. The factor was applied as follows:

$$\textit{Concentration at Scarboro } \left(\frac{mg}{m^3}\right) \text{ ' } \textit{Annual Average Release Rate } \left(\frac{mg}{s}\right) \times \textit{Empirical } \chi/Q \left(\frac{s}{m^3}\right) \qquad (7.11)$$

For application to the Task 2 assessment, the empirical χ/Q factor was specified as a PDF in the form of a custom distribution, incorporating each of the 20 empirical χ/Q's estimated using the uranium data. The factor was applied to each annual estimate of direct airborne releases from the Y-12 Plant for 1953 to 1962. To convert annual release estimates to average release rates, totals were assumed to be evenly distributed over the year in question.

Airborne concentrations resulting from cross-ridge travel were added to the Scarboro concentrations resulting from mercury volatilization from EFPC (described in Section 7.3). The relative contributions of Y-12 Plant emissions and emissions from EFPC to air concentrations estimated at the Scarboro Community for 1953 to 1962 are shown in Figure 7-2. For 1950-1952 and 1963-1990, when major airborne mercury releases from Y-12 did not occur, airborne mercury concentrations at Scarboro were estimated based on volatilization from EFPC alone.

7.3 Mercury Concentrations in Air Due to Volatilization from EFPC

In 1993, Ralph Turner of ORNL and Nicolas Bloom of Frontier Geosciences measured elevated mercury concentrations in tree rings of red cedars growing in the EFPC floodplain (Turner and Bloom 1995). Several investigators have measured elevated mercury concentrations in tree rings from areas with elevated airborne mercury concentrations, such as near chloralkali plants (Lodenius et al. 1994; Turner and Bloom 1995). The tree ring mercury was assumed to come from foliar uptake of airborne mercury, because plants

take up (and release) mercury through their foliage and uptake of mercury through tree roots is minimal (Beauford and Barringer 1977; de Temmerman et al. 1986; Mosbaek et al. 1988; Lindberg et al. 1995).

Trees add a new ring for each year of growth. Therefore, mercury concentrations in tree ring core samples can provide an indication of historical trends in airborne mercury concentrations (Lodenius 1990; Turner and Bloom, n.d; Turner and Bloom 1995). Indeed, tree ring data from the EFPC floodplain trees, plotted in Figure 7-3, show that concentrations of mercury in tree rings corresponding to the ten years (1953-1962) surrounding the period of peak mercury releases from Y-12 (1955-1959) were considerably higher than tree ring concentrations for earlier or later periods.

As discussed in Section 5.2, some volatilization of elemental mercury to form mercury vapor occurs at room temperature. Mercury may volatilize from surface water and soil (Lindberg et al. 1991; Xiao et al. 1991; Lindberg et al. 1995; Lindberg et al. 1996). Because exchange of air between Y-12 and the EFPC floodplain is impeded by Pine Ridge (U.S. Weather Bureau 1953; Gifford 1995), the source of mercury in EFPC trees is likely primarily volatilization from EFPC and not direct air emissions from Y-12.

Tree ring data clearly indicate that air concentrations of mercury in the floodplain were elevated during the years of peak releases from Y-12. However, with present knowledge, mercury concentrations in individual tree rings cannot be used to reliably estimate annual average airborne mercury concentrations at the tree locations. This is because mercury is relatively mobile in the sapwood of the tree and can move from ring to ring before the sapwood becomes heartwood. In addition, individual trees appear to respond quite differently to airborne mercury. For example, Figure 7-4 shows mercury concentrations (in $\mu g\ g^{-1}$) in tree rings from two trees that grew very close together in the EFPC floodplain. Both of these trees experienced the same temporal pattern of exposure to airborne mercury, peaking in 1957, but it is clear from Figure 7-4 that the two trees responded very differently to the airborne mercury exposure. The difference may be due to differences in the rate of uptake of mercury and translocation of mercury between tree rings, between the two trees, and from year to year within a single tree. The lack of a direct correspondence between tree ring data and annual average airborne mercury concentration is discussed in Appendix O. At this time, there is not enough information on mercury uptake by red cedars, translocation of mercury within the trees, and variation from tree to tree and from year to year within a single tree to allow detailed mathematical modeling of mercury in tree rings as a function of ambient airborne mercury concentrations.

Historical airborne mercury concentrations in and near the EFPC floodplain were estimated by modeling mercury volatilization from EFPC. The modeling was based on the estimated fraction of mercury discharged from Y-12 to EFPC that subsequently escaped into the air above the EFPC floodplain between Y-12 and the junction of EFPC with Poplar Creek.

Figure 7-2: Contributions to Calculated Average Airborne Mercury Concentrations at the Scarboro Community

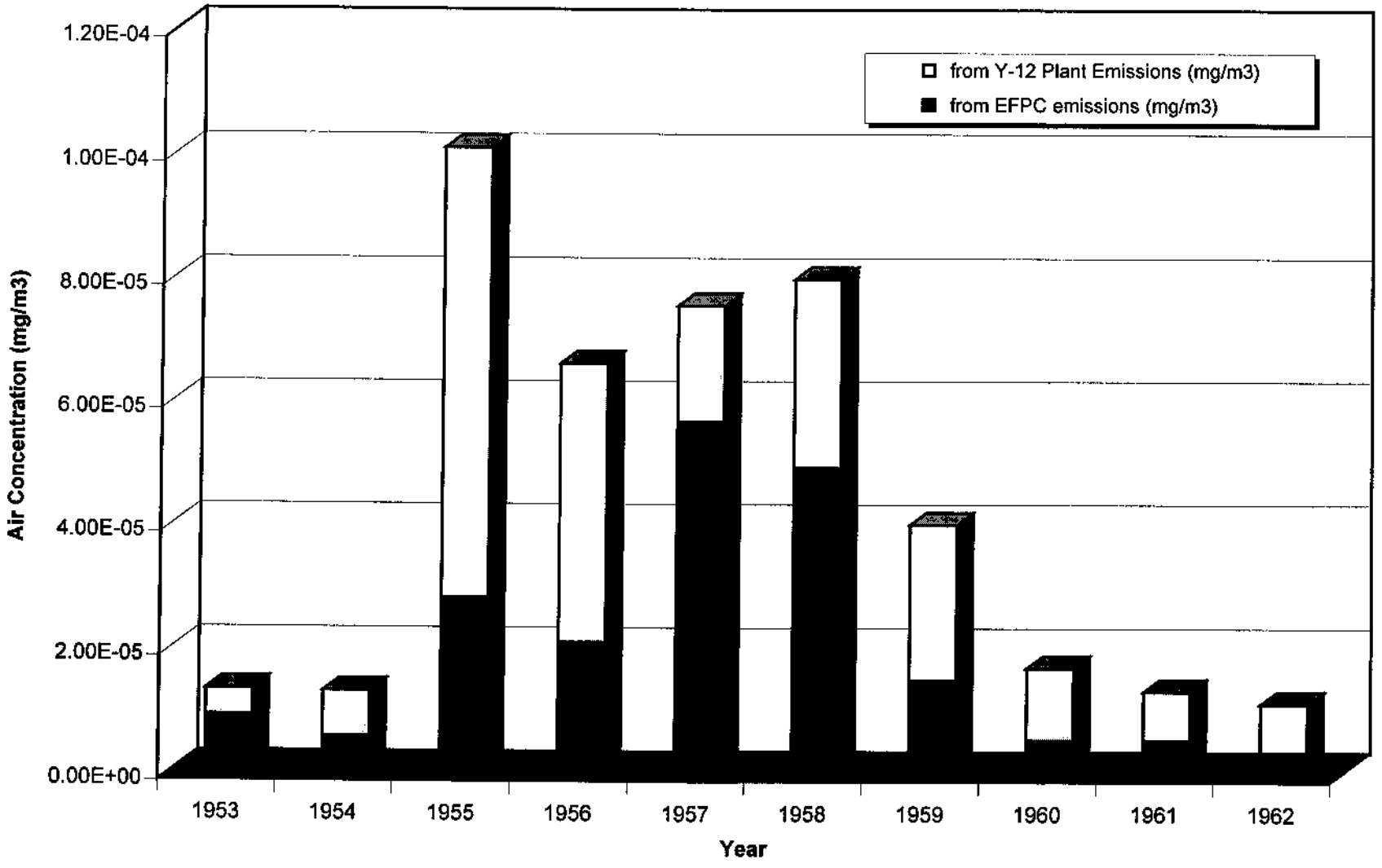


Figure 7-3: Relationship of Mercury Concentrations Measured in Rings of Trees Along EFPC (mg kg^{-1}) toY-12 Releases to EFPC (mg L^{-1})

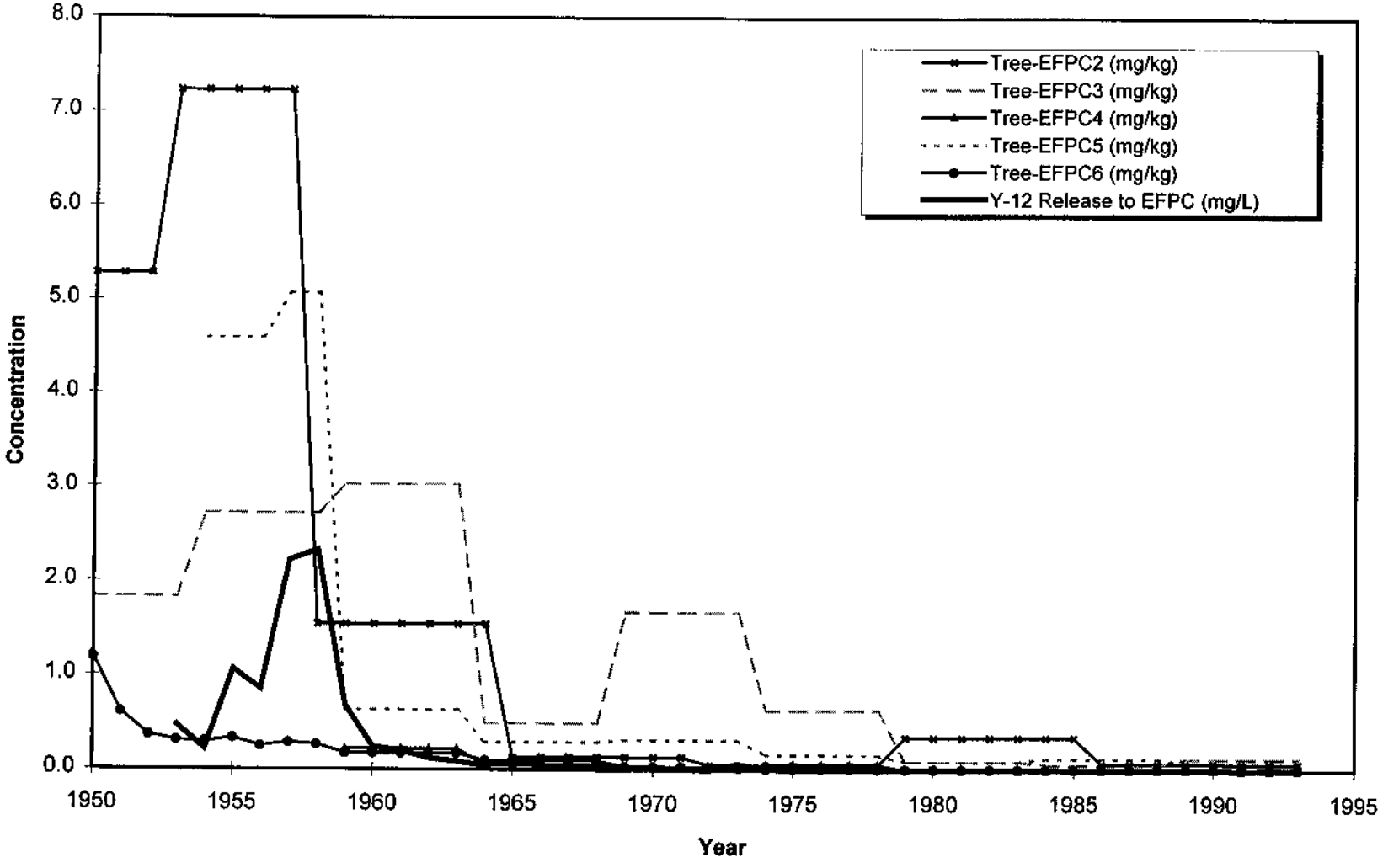
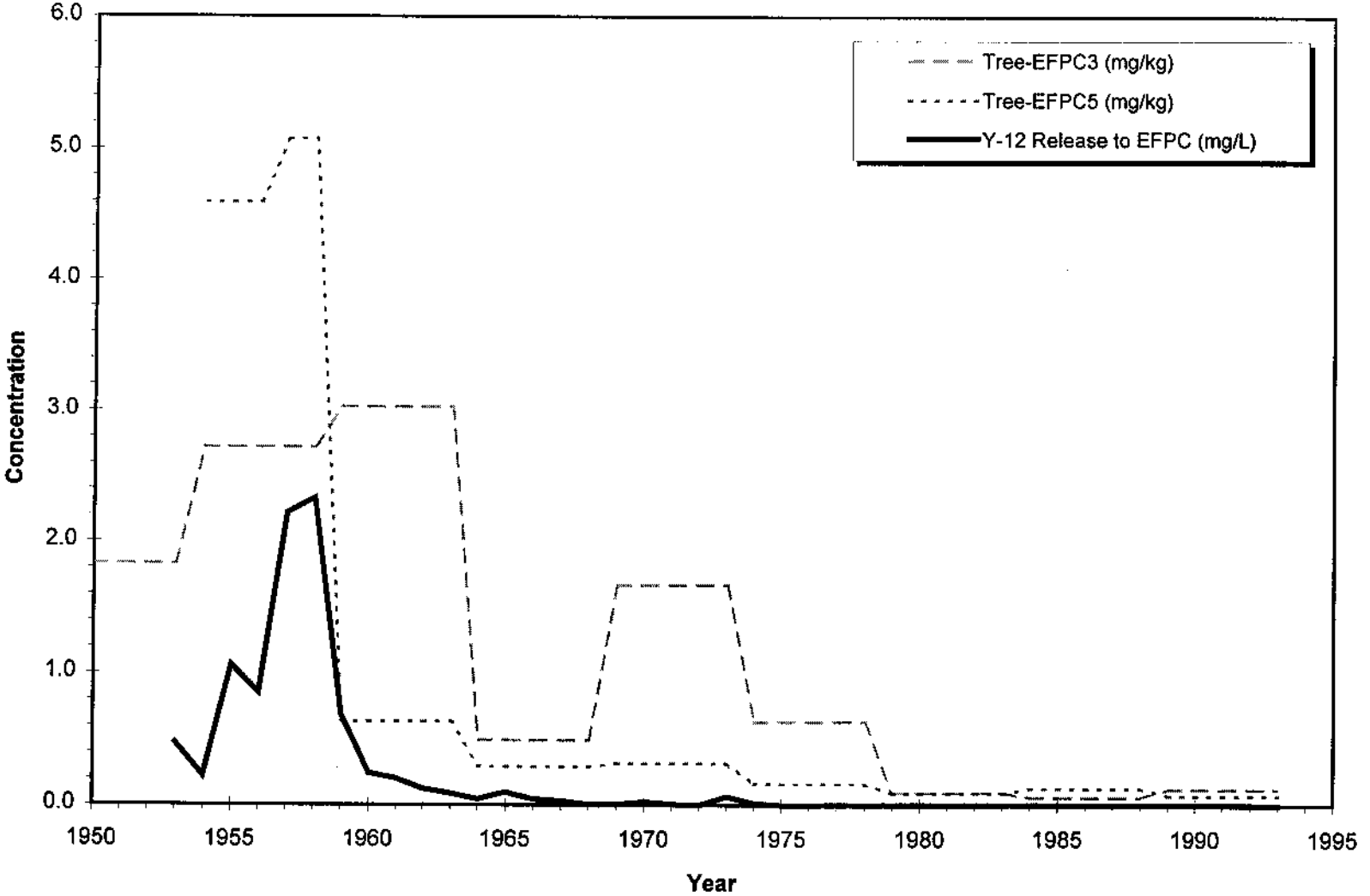


Figure 7-4: Mercury Concentrations Measured in Tree Rings from Tree-EFPC3 and Tree-EFPC5 (µg/g)



Data collected by Lindberg et al. (1995) in the EFPC floodplain can be used to assess the possible significance of soil emission of airborne mercury during the peak emission years, compared to emissions from EFPC. Peak air concentrations estimated from air dispersion modeling for EFPC were about 1.7 µg m^{-3} (1,700 ng m^{-3}) immediately adjacent to the creek. In contrast, Lindberg et al. measured an average air concentration in the floodplain of about 5 ng m^{-3} in 1993 when the wind blew from the southeast over soil contaminated with mercury at an average level of about 40 mg kg^{-1}. This is approximately equal to the peak soil contamination used in the dose reconstruction. So, the contribution to air contamination from soil emission is likely to have been minimal in comparison to the release from EFPC.

The air modeling approach and the modeling results are summarized below.

7.3.1 Approach and Assumptions

Emission of elemental mercury vapor from EFPC was estimated based on:

- Annual releases of mercury from Y-12 to EFPC, and
- Assumptions about the fraction *f* of the total mercury released from Y-12 that volatilized as the water traveled from Y-12 to the junction between EFPC and Poplar Creek.

Mercury volatilization from EFPC was modeled assuming the annual average loss of mercury to air from a one meter stretch of EFPC is proportional to the annual average mass of mercury in that stretch of the creek. This approach assumes that loss from narrow fast-moving sections is equivalent to loss from wide slow-moving sections.

7.3.2 Model Description

Modeling of mercury volatilization from EFPC was comprised of two steps: source modeling of mercury volatilization from EFPC, and dispersion modeling of mercury emitted from the creek to receptor locations. Each of these steps is described in the following sections.

Source Modeling

The mass M_l of mercury in EFPC at a distance l from Y-12 can be approximated using the following equation:

$$M_l = M_0 \, e^{\frac{\ln (1 - f)}{L} l} \qquad (7.12)$$

where:

M_l = Mass of mercury at distance l from Y-12 (g y^{-1})

M_0 = Initial mass of mercury released to EFPC from Y-12 (g y^{-1})

L = Total length (in meters) of EFPC (assumed to be 23,200 m)

l = Distance of start of segment from discharge point at Y-12 (m)

f = Fraction of the mercury in the original discharge lost to air as the water flows from Y-12 to the junction with Poplar Creek

For EFPC air dispersion modeling, EFPC was divided into a total of 403 straight-line segments, each characterized by a distinct length, d, with a maximum length of 100 meters.

The total annual mass lost or emitted from each segment i, in g y^{-1}, can then be calculated as follows:

$$M_{emitted,i} = M(l_i) - M(l_{i+1}) \quad (7.13)$$

where:

l_i = Distance from Y-12 to beginning of segment i

l_{i+1} = Distance from Y-12 to beginning of segment i+1

Derivation of this equation is described in Appendix P. Assumptions used to characterize the parameters in the source model are described below.

Initial Mass of Mercury Released to EFPC from Y-12 (M_0)

The initial mass of mercury released to EFPC from Y-12, M_0, was based on data for 1950 to 1990 on pounds of mercury released to EFPC per year, as described in Section 4.5.

Fraction of Mercury Lost to Air (f)

Comparison of data describing mercury concentrations in EFPC floodplain soil and Watts Bar Reservoir sediment to pounds of mercury released to EFPC per year from Y-12 and the limited measurements of mercury concentrations in EFPC surface water near the junction of EFPC and Poplar Creek, as described in Section 7.1, suggest that the fraction of the mercury released to EFPC that volatilized from the creek may have been relatively small. Comparison of surface water concentrations at Y-12 vs. near the EFPC/ Poplar

Creek junction in 1956-1961 suggests that, on average, about 60 to 90% (range 40-95%) of the mercury released to EFPC was "lost" between Y-12 and the junction. Measurements of mercury concentrations in floodplain soil by TVA, however, suggest that about 60% of the total mercury released from Y-12 was still in the floodplain in 1984**S** additional mercury that was initially bound to sediments was probably washed downstream during subsequent flood events.

The concentration of volatile mercury in air immediately above the water surface has been directly related to the concentration of dissolved gaseous mercury (DGM) in the water body. DGM is primarily dissolved elemental mercury (Hg^0). Because of its low solubility (6×10^{-6} g/100 ml water at 25 C) and high vapor pressure (Henry's Law constant =0.3), Hg^0 is very volatile. Thus, formation of DGM favors the removal of mercury from the system through evasion of mercury vapor.

Limited information is available quantitatively describing volatilization of mercury from surface waters bodies. Review of the scientific literature and discussions with experts indicate that most data that are available data are from lakes and ponds; data from moving water bodies such as streams or rivers are extremely limited. However, DGM has been measured in EFPC. Measurements in EFPC below Reality Lake/New Hope Pond in 1996 showed that DGM concentrations ranged from 0.016% to 0.21% (average 0.095%) of total mercury concentrations (Turner personal communication). DGM concentrations in the upper creek (above Reality Lake/ New Hope Pond) ranged from <1% to 3.3% of total mercury (Saouter et al 1995, Liebert et al 1991). The percent DGM measured in EFPC is consistent with DGM values measured in pristine and uncontaminated systems. For example, data presented by Harris and Snodgrass (1997) suggest DGM in a Canadian Shield Lake is 1.3 to 2.3% of dissolved mercury, or <1% of total mercury.

While DGM reflects the fraction of the total mercury in a water body that is in a volatile form, the actual fraction of the total mercury that volatilizes over a span of time may be larger or smaller, depending on such factors as the turbulence of the water and the time and distance over which the water travels. Also, as elemental mercury is removed from the water by evasion, more elemental mercury will be formed as the equilibrium among reactants and products drives the reactions that produce elemental mercury (e.g., reduction of divalent mercury (Hg^{2+}) by biotic and abiotic processes) to produce more elemental mercury.

Measurements of mercury in EFPC in 1996 showed that most of the mercury in the EFPC water column is adsorbed to suspended particulates (Sauoter et al. 1995). In general, the percentage of mercury in the water column that was present as dissolved mercury during 1989 and 1996 decreased with downstream distance: the percentage of mercury present as dissolved mercury was 68% at EFPCM 15.7 (above Reality Lake/ New Hope Pond), 16% at EFPCM 12.9, and 2.1% at EFPCM 1.3. Mercury that is adsorbed to particulate is not very available, either to volatilize or to be methylated. However, over the length of EFPC, the percent of mercury present as DGM relative to total dissolved mercury was generally consistent (around 1.0 to 1.5% of total dissolved mercury). This suggests that with increasing distance downstream, less of the mercury in the water column is available to be reduced and evade to air as more and more of the mercury binds to suspended particulate.

Studies of the "reductive potential" of water in EFPC and other sites (Barkay et al. 1992, Turner et al. 1993, Saouter et al. 1995) demonstrate that a number of water quality parameters can decrease or enhance the reduction of instream mercury to DGM. Parameters that inhibit reduction of waterborne mercury include low pH and high suspended matter concentrations. Suspended matter, particularly minerals and dead organic matter, compete strongly with the reduction process for mercury in the solution phase and, once sorbed to suspended matter, mercury does not appear to be available for reduction or the rate of reduction is much slower than for dissolved mercury (Turner et al. 1993). It is difficult to determine whether water quality conditions in EFPC in the 1950s consistently favored or inhibited instream reduction of mercury. However, it may be reasonably assumed that total suspended solids concentrations were high in EFPC during the 1950s (having an inhibitory effect on reduction) and that pH fluctuated widely between values favorable to reduction (high pH) and inhibitory to reduction (low pH). Physical parameters such as temperature, wind speed, and mixing of the water column, can also affect the rate of evasion of elemental mercury from the water column (Saouter et al. 1995).

In a study conducted by Saouter et al. (1995), a laboratory microcosm consisting of water, sediment, and air compartments was used to simulate mercury geochemical cycling in Reality Lake. Total and dissolved mercury, total gaseous mercury, and methylmercury were measured in the water at the inlet and outlet of the microcosm, and at the inlet and outlet of Reality Lake. Results indicate that in the microcosm, approximately 32% of the total mercury in water was "lost" between the inlet and the outlet. In Reality Lake, approximately 7.4% of the mercury was "lost". However, measurements of mercury concentrations in sediment at the bottom of the microcosm indicate that most of the "lost" mercury adsorbed to particulates and settled out of the system— only about 2 to 7% of the mercury that was lost appeared to have volatilized.

Based on the above considerations, it was assumed that:

- On an annual average basis, between 60 and 90% of the mercury released from Y-12 was "lost" between Y-12 and the junction.
- On an annual average basis, 60% or more of the mercury that was released from Y-12 was bound to sediments in the floodplain, suggesting that the dominant process removing mercury from surface water was likely absorption to particulates and sedimentation.
- On an annual average basis, the likely maximum amount of mercury released from Y-12 that volatilized from EFPC was 30%.

Based on these considerations, the upper-bound estimate of the fraction of mercury discharged from Y-12 in a given year that escaped to air above EFPC was assumed to be 30%. The lower-bound was assumed to be 1%, based on the assumption that almost all of the mercury was lost to sediment. The best-estimate was assumed to be 5%, determined by multiplying the mid-range of the estimated percentage of mercury lost to either air or sediment between Y-12 by the mid-range of the estimated percentage of "lost" mercury assumed to have volatilized in the microcosm and Lake Reality studies.

Dispersion Modeling

Air dispersion modeling was conducted to estimate ground-level concentrations of airborne mercury at receptors near EFPC, using estimates of emission rates from EFPC. The most recent version of the USEPA-approved Industrial Source Complex Short Term (ISCST3) computer model was used (USEPA 1995c, Version 96113). ISCST3 is a Gaussian air dispersion model that calculates ground-level concentrations downwind from an area source from the following double integral in the upwind (*x*) and crosswind (*y*) directions:

$$\chi = \frac{Q_A K}{2\, pi\, u_s} \int_x \frac{V D}{\sigma_y \sigma_z} \int_y \exp\left[-0.5\left(\frac{y}{\sigma_y}\right)^2\right] dy\, dx \qquad (7.14)$$

where:

Q_A	=	Area source emission rate (mass per unit area per unit time)
K	=	Units scaling coefficient
V	=	Vertical term
D	=	Decay term as a function of x
σ_y , σ_z	=	Standard deviation of lateral and vertical concentration distribution (m)
u_s	=	Mean wind speed (m s^{-1}) at release height

The dispersion modeling used unit emissions (1 g s^{-1}) from each creek segment. This determines the contribution to annual average airborne concentration at each receptor from a unit release by each segment. The contribution to the annual average air concentration at each receptor from a given segment is then obtained by multiplying the contribution from a unit release at the segment by the estimated emission rate (*Q*) from that segment for each year of emission. The total annual average airborne concentration at each receptor for each year is calculated by summing the contributions from all segments.

Required inputs to the air dispersion model included:

- Location, length, width, and orientation of area sources used to represent EFPC
- Emission rates for each source
- Meteorological data representative of conditions in the EFPC floodplain
- Receptor locations

EFPC was represented as a series of 403 elongated area sources along the creek with a maximum length of 100 meters and a nominal width of 15 meters. The length of each segment was chosen to approximate the shape of the creek, as shown in Figure 7-5. Source parameters necessary to characterize emissions for air dispersion modeling, including length, width, orientation and emission rate for each segment are presented in Appendix P (see Tables P-1 and P-2). Three values of the mercury loss fraction *f* (0.01, 0.05, 0.3) were modeled.

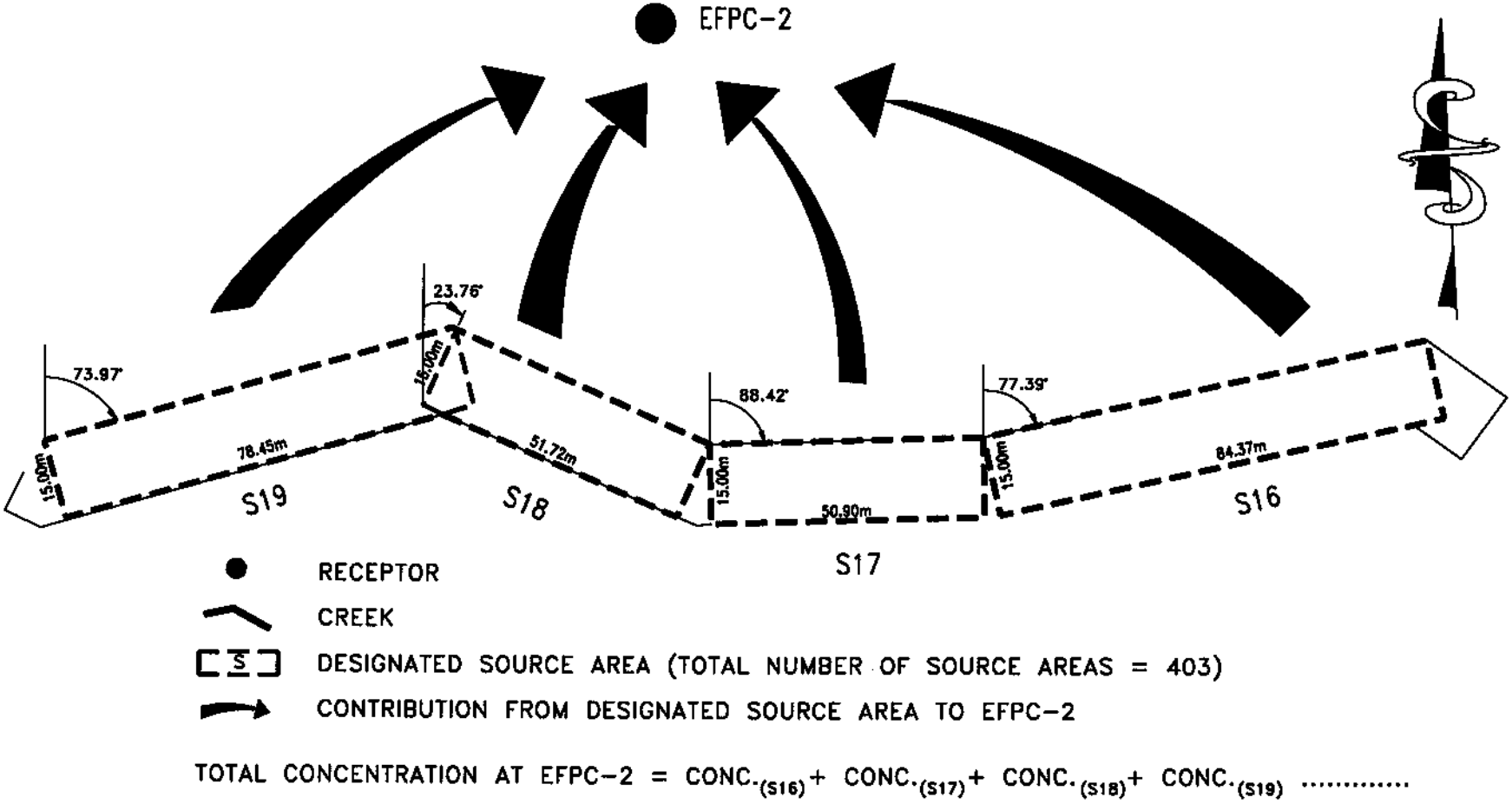


Figure 7-5: Conceptual Model for Mercury Releases from EFPC Using the ISCST3 Air Dispersion Model

ChemRisk® A Division of McLaren/Hart

OAK1997/EPPC_REC/FIG7-4.DWG 7/9/97 vlb

EFPC is in a valley between two ridges– Blackoak Ridge to the northwest and Pine Ridge to the southeast. Since the EFPC floodplain is generally flat, ISCST3 can be used to model air dispersion near the creek. The two ridges create a wind pattern that is mainly in the northeast-southwest direction. During the years of greatest air emissions of mercury (i.e., 1950-1963), hourly meteorological data for the EFPC floodplain are not available. Monthly average data from the Oak Ridge town center station (Station 886) were compared to hourly average data collected from 1987-1992 at the Y-12 East Meteorological station. Based on this comparison, meteorological data from the Y-12 East station for the year 1987 were used to provide hourly wind speed, wind direction, temperature, stability class, and mixing height information to model releases from EFPC. Receptor locations modeled near EFPC include the Scarboro Community, Robertsville School, the EFPC farm family, the community receptors, and the locations of trees in the EFPC floodplain analyzed for mercury content in their tree rings (Figures 1-2 and 7-6).

7.3.3 Simulation Results and Uncertainty in Exposure Estimates

In the Monte Carlo uncertainty analysis, air concentrations at receptors for a given year due to volatilization of mercury from EFPC were characterized by defining a distribution of air concentrations at the receptors based on the bounding of volatilization rates of mercury from EFPC (i.e., assuming 1%, 5%, and 30% of mercury volatilizes from EFPC over the length of the creek) and uncertainties induced by the modeling approach:

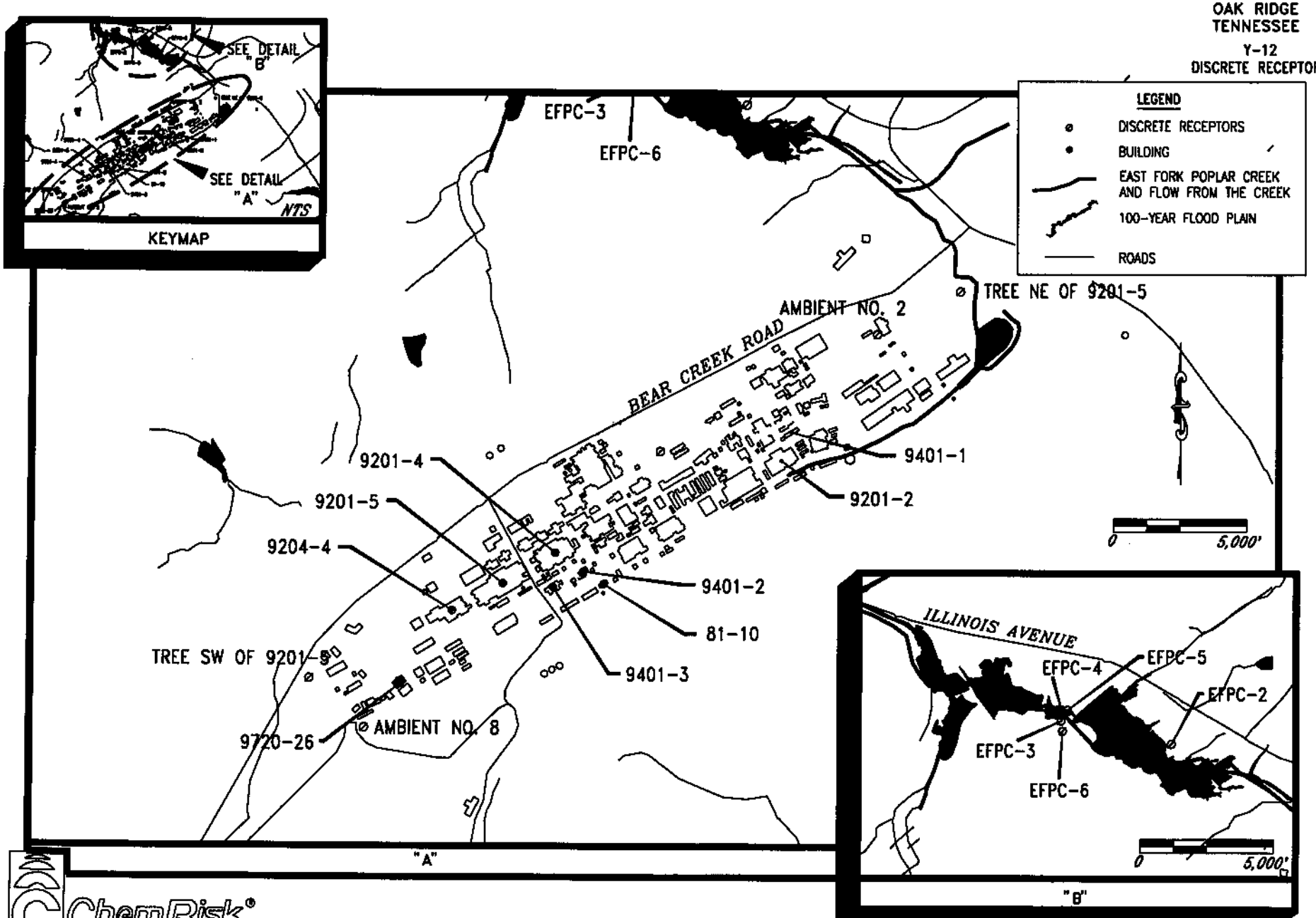
$$C_{air\&vol} \text{ ' } C_{air\&vol(\mathrm{mod})\ (1\%,\ 5\%,\ 30\%)} \times M_{Unc} \qquad (7.15)$$

where:

$C_{air\text{-}vol}$	=	Receptor-specific air concentration due to volatilization of mercury from EFPC, used in Monte Carlo uncertainty analysis (µg m^{-3})
$C_{air\text{-}vol(mod)}$	=	Modeled receptor-specific air concentration due to volatilization of mercury from EFPC (µg m^{-3})
M_{unc}	=	Uncertainty in air dispersion model (unitless)

PDFs used in the calculations to characterize air concentrations for each year due to volatilization of mercury from EFPC are summarized in Table 7-11. The PDF used to characterize ISCST3 model uncertainty for the EFPC area was again represented by a lognormal distribution with geometric mean of 1 and geometric standard deviation = 1.4 (i.e., 95% of the distribution is within a factor of 2 of the geometric mean), as described in Section 7.2.3.

FIGURE 7-6
OAK RIDGE
TENNESSEE
Y-12
DISCRETE RECEPTORS



ChemRisk®
A Division of McLaren/Hart
10755 VON KARMAN AVENUE, IRVINE, CA 92714

Oakridge/Fig1-2A.dwg (7/9/97) vm

Table 7-11: Probability Density Functions for Air Concentrations Due to Volatilization from EFPC

Parameter	PDF	
	Distribution Type	**Description**
Modeled air concentration ($C_{air\text{-}vol\ (mod)}$)	Logtriangular	Lower-bound = Modeled air concentration at 1% emission Best estimate = Modeled air concentration at 5% emission Upper-bound = Modeled air concentration at 30% emission
Model uncertainty (M_{Unc})	Lognormal	Geometric mean = 1 Geometric SD = 1.4

7.4 Mercury Concentrations in Soil and Sediment

Investigations of mercury concentrations in soil in the EFPC floodplain have shown that soil concentrations are elevated due to sedimentation of mercury from water and deposition of airborne mercury. As discussed in Sections 7.2 and 7.3, a large percentage of the mercury released from Y-12 to EFPC was likely bound to sediments in EFPC or the EFPC floodplain. It is likely that mercury concentrations in surface soil in the floodplain were highest during the period when releases from Y-12 were highest. Later surface soil concentrations were probably lower due to removal of soil during flood events and deposition of materials with lower concentrations. Numerous floods have been reported on EFPC over the last 200 years. Floods occur on the creek at an average of approximately four per year (TVA 1959). During the period of peak releases from Y-12, a large flood occurred in April 1956 (TVA 1959).

As discussed in Section 6.1.3, pathways of exposure to mercury in floodplain soil and sediment could have included direct contact (e.g., ingestion and dermal contact) as well as indirect pathways from ingestion of vegetables grown in or near the floodplain or ingestion of milk or meat from animals that grazed in the floodplain and ingested soil.

The earliest measurements of mercury in EFPC sediment were three sediment ("mud") samples collected by Sanders in 1970 (Sanders 1970). The samples were collected just below New Hope Pond (0.90 mg kg^{-1}), in EFPC near Wiltshire Estate (1.6 mg kg^{-1}), and in EFPC near the Oak Ridge Country Club (11.3 mg kg^{-1}). No further sampling was conducted until 1982 when Van Winkle et al. (1984) collected a total of seven surface sediment samples along the length of EFPC. Concentrations ranged from 19 to 127 mg kg^{-1}, with the maximum concentration measured at EFPC Mile 13.8, just downstream from New Hope Pond. Van Winkle et al. (1984) also collected a sediment core from New Hope Pond to gauge mercury deposition since dredging of New Hope Pond in 1973. Data from this core show a peak sediment

concentration at a depth of 30 to 45 inches below the surface, indicating that mercury releases from Y-12 had decreased since 1973.

The earliest measurements of mercury in soil in the EFPC floodplain were made in 1983. The Oak Ridge Associated Universities (ORAU) conducted extensive sampling of mercury concentrations in EFPC floodplain soil and elsewhere in the Oak Ridge community between 1983 and 1987 as part of the Oak Ridge Task Force (ORTF) investigations (MMES 1984, 1985, 1986, 1987, 1988; TDHE 1983; Hibbitts 1984, 1986). The TVA also conducted an extensive sampling program for mercury in the EFPC floodplain as part of the Instream Contaminant Study, also a part of the ORTF investigations (TVA 1985b). The TVA collected 122 sediment cores from the EFPC floodplain, 19 instream sediment samples from EFPC, and 15 sediment cores from the Clinch River and Watts Bar Reservoir.

Another extensive sampling program for mercury in EFPC floodplain soils was conducted between 1990 and 1992 by SAIC, as part of the EFPC Floodplain Remedial Investigation (SAIC 1994b). SAIC conducted the sampling in two phases. During Phase Ia, they collected soil samples in the floodplain from three areas of known contamination (NOAA, Bruner's Center sites, and Sturm sites). During Phase Ib, they conducted large scale sampling of floodplain soils in 159 transects across the floodplain and at right angles to the creek at 100 m (330 ft) intervals. Almost all of these samples were collected over three depth intervals: 0 – 16 inches below ground surface (bgs), 16 – 32 inches bgs, and 32 – 48 inches bgs. Samples from each depth interval were homogenized (blended into a uniform mixture) for analysis.

As discussed in Section 5.2 and described in greater detail in Appendix K, a number of investigations using several different extraction and analytical methods were conducted from 1988 through 1994 to attempt to speciate the mercury in floodplain soil (Barnett and Turner 1995; SAIC 1994b; Revis et al. 1989). Results suggest that much of the mercury is currently present in forms not likely to be mobile in soil. Revis et al. (1989) reported that mercury in soils in the EFPC floodplain were primarily restricted to the top 40 cm (about 16 inches) of the soils (Henke et al. 1993). Movement of elemental mercury vapor and mercuric mercury (Hg^{2+}) ions in soils and sediments is frequently hindered by the tendency of mercury to bind to natural organic matter and, to a lesser extent, to fine-grained minerals (Henke et al. 1993). In addition, the solubility of elemental mercury is low, limiting its potential to leach through soil (Porcella 1994). Based on these data, it is assumed that mercury deposited historically on EFPC floodplain soils did not significantly leach into deeper soils. This assumption is borne out by the results of the SAIC sampling, which indicate that the average concentration measured in the top depth interval (0 – 16 inches below ground surface (bgs)) generally exceeded average concentrations in the deeper depth intervals (16 – 32 inches bgs and 32 – 48 inches bgs). For example, at the location in the floodplain near that assumed to have been occupied by the EFPC farm family (e.g., between approximately EFPC Mile 9.5 and 10.5), the arithmetic mean concentration in the top depth interval was 13 mg kg^{-1}, while the arithmetic mean concentration in the second interval was 5.0 mg kg^{-1}.

The exposure point concentrations used to evaluate exposures to mercury in floodplain soil and EFPC sediment are described below.

The project team used soil data collected by SAIC between July 1991 and May 1992, during Phase Ib of the EFPC Floodplain Remedial Investigation (RI), to estimate exposures to mercury in floodplain soil and sediment at the EFPC Floodplain Farm Family and Robertsville School population locations. Data from this sampling program were selected for use in evaluating exposures to individuals living near or recreating in the floodplain because this is the most extensive data set available. Exposures of Scarboro Community residents to mercury in surface soil were evaluated using soil data collected in the Scarboro area by ORAU in 1984, because no soil samples were collected in the Scarboro Community during the 1991-92 EFPC RI.

As discussed, samples were collected during the EFPC RI from 159 transects extending across the floodplain, spaced 100 m apart along 23 km of the creek. Soil samples were taken at the edge of the creek and every 20 m (65 ft) away until the elevation of the 100-yr floodplain was reached. Samples were collected to a maximum depth of 48 inches at depth intervals from 0 – 16 inches below ground surface (bgs), 16 – 32 inches bgs, and 32 – 48 inches bgs. Within these depth intervals, samples were homogenized, such that reported concentrations represent the average mercury concentration over the entire depth interval. In general, of the three depth intervals, homogenized samples from the top (0 – 16 inches) interval had the highest concentration. This interval likely reflects mercury deposited on the soil during the period of highest releases from Y-12. Samples from this interval were used to characterize exposures to soil and sediment in the floodplain.

To characterize the stratification of mercury concentrations in soil with depth, SAIC conducted a Vertical Integration Study in 1992 during which they collected five 18-inch long soil cores and divided each core into 1-inch depth intervals. Cores were collected from four locations: the EFPC/ Poplar Creek confluence, Grand Cove Subdivision, the Bruners Study site (two samples), and the NOAA Study Site. Data from this investigation indicate that averaging of individual results from 1-inch samples collected from 0 to 16 inches bgs produced concentrations that are generally consistent with concentrations in homogenized 0 – 16 inch depth interval samples from the same location (Table 7-12). Results also showed that the soil horizon containing the highest mercury concentrations was typically buried under 9 to 16 inches of overburden; this deeper horizon probably represent mercury deposited during the period of highest releases from 1955 – 1959. Concentrations in this horizon were about 300% to 500% greater than concentrations averaged over the 0 **S** 16 inch interval. In contrast, concentrations measured in depth intervals closer to the surface were as low as <1% of the concentration averaged over the 0 **S** 16 inch interval.

To adjust for stratification with depth, and the likelihood that surface soil concentrations in the past were higher than the present, average soil concentrations measured in the 0 **S** 16 inch depth interval samples were adjusted based on the mass of mercury annually released from Y-12 between 1950 and 1990.

Table 7-12: Results of the SAIC (1994) Vertical Integration Study of Soils in the EFPC Floodplain

Depth Interval	NOAA (#N334E02)		Bruners Center (#E534N00)		Bruners Center (#E534N00-Dup)		EFPC/ Poplar Creek (#E203N00)	
	Conc. (mg kg^{-1})	% of Homogenized	Conc. (mg kg^{-1})	% of Homogenized	Conc. (mg kg^{-1})	% of Homogenized	Conc. (mg kg^{-1})	% of Homogenized
0-1 in.	451	34%	<1.7	0.090%	NA	NA	<3.9	1.1%
1-2 in.	593	45%	25	2.8%	29	3.2%	14	7.9%
2-3 in.	642	49%	<2.1	0.11%	8.1	0.90%	<4.0	1.1%
3-4 in.	647	49%	32	3.6%	71	7.9%	5.4	3.1%
4-5 in.	NA	NA	40	4.5%	1990	220%	6.5	3.7%
5-6 in.	NA	NA	492	55%	<3.6	0.20%	NA	NA
6-7 in.	510	39%	1740	190%	<3.6	0.20%	NA	NA
7-8 in.	1190	90%	2450	270%	<3.3	0.18%	14	7.9%
8-9 in.	1920	150%	2440	270%	<3.3	0.18%	63	36%
9-10 in.	2870	220%	2740	310%	<3.7	0.21%	307	170%
10-11 in.	1950	150%	3420	380%	<4.2	0.23%	425	240%
11-12 in.	NA	NA	1870	210%	226	25%	228	130%
12-13 in.	NA	NA	292	33%	181	20%	117	66%
13-14 in.	318	24%	374	42%	2080	230%	96	54%
14-15 in.	306	23%	295	33%	2920	330%	56	32%
15-16 in.	436	33%	168	19%	2130	240%	45	25%
Average	986		1024		643		99	
Concentration in Homogenized Sample								
0-16 in.	**1320**		**898**		**898**		**177**	

NA Data not available for this depth interval**S** Sample result was rejected.

Surface soil concentration PDFs used to estimate dose were characterized as follows:

$$C_{soil} = C_{soil\&measured} \times AF_{depth} \quad (7.16)$$

where:

C_{soil} = Soil concentration used in Monte Carlo uncertainty analysis

$C_{soil\text{-}measured}$ = Average soil concentration measured in cores (0 – 16 inch depth interval) by SAIC during the EFPC RI (EFPC farm family and Robertsville School children) or in the surface depth interval by ORAU (Scarboro Community), for each area of interest

AF_{depth} = Adjustment factor, applied to floodplain soil/sediment samples only (to reflect stratification of mercury concentrations at depth and the likelihood that past surface soil concentrations were different than concentrations in the homogenized sample)– adjustment factors were not applied to the Scarboro soil data set, since these samples were collected in the surface interval (0 to 3 inches bgs) and soils in this area were not subject to EFPC flooding

The input parameters are described below.

Soil Data

Soil data collected in the floodplain during the EFPC RI or in the Scarboro Community by ORAU were used to evaluate exposures to individuals who resided near or recreated in the floodplain. Subsets of data from different areas were used to evaluate exposures via different pathways as follows:

- For the EFPC Farm Family and Robertsville School children populations, direct contact exposures to soil (such as ingestion or dermal contact with soil) were evaluated using soil data collected during the EFPC RI across the width of the floodplain. The same data set was used to evaluate the "ingestion of soil by livestock" pathways for the EFPC Farm Family. For the Scarboro population, direct contact exposures to soil were characterized using surface soil data collected in and near the Scarboro Community by ORAU– these data were collected from approximately 0 to 3 inches bgs.

- For the EFPC Farm Family population, uptake of mercury from soil into vegetables was evaluated using soil data collected during the EFPC RI in an area spanning between 20 m from the creek and the edge of the 100-year floodplain. It was assumed that the frequency of inundation of lower elevations by floodwater precluded growing vegetable gardens in these areas. For the Scarboro population, uptake of mercury from soil into vegetables was evaluated using surface soil data collected in the Scarboro community by ORAU.

- Sediment samples collected in EFPC were limited. Therefore, exposures to mercury in instream sediment by members of the Scarboro community, the EFPC Farm Family population, and Robertsville School children were evaluated using soil samples collected on the edge of EFPC during the EFPC RI.

Soil data used to characterize exposures to each population through different pathways are tabulated in Appendix Q.

Adjustment Factors

Soil concentration adjustment factors applied to samples collected in the EFPC floodplain were derived assuming that:

- During the peak release years, concentrations at the surface were significantly *higher* than average concentrations over the entire depth interval,

- During recent years, concentrations at the surface were significantly *lower* than average concentrations over the entire depth interval, and

- During interim periods, concentrations at the surface were *at or near* average concentrations over the entire depth interval.

Historical concentrations in floodplain surface soils were estimated by applying adjustment factors to the concentrations reported for homogenized samples from the 0 **S** 16 inch depth interval. It was assumed that the mercury concentrations in floodplain surface soil between 1950 and 1990 changed in proportion to the pattern of mercury releases from Y-12. Thus, it was assumed that surface soil concentrations was highest during 1957, when the mass of mercury released was the highest, and lowest during 1990.

Evaluation of the floodplain soil data from the SAIC Vertical Integration Study (Table 7-12) shows that the highest soil concentrations were measured at 10 to 15 inches bgs. Assuming that the highest concentrations were deposited during 1957, this suggests that the rate of deposition of sedimentary materials in the floodplain ranged between ¼ and ½ inch per year, with an average closer to ¼ inch per year in three of the four stratified samples (NOAA, Bruner's Center, and EFPC/ Poplar Creek junction). Assuming a depositional rate of ¼ inch per year, soil layers in the stratified samples were "dated", and

adjustment factor ranges calculated based on the relationship between the concentration measured in a "dated" depth interval, and the concentration measured in the corresponding homogenized sample (Table 7-13). To estimate surface soil concentrations corresponding to different years, these adjustment factors were applied to soil concentrations measured in homogenized samples collected during the EFPC RI from the 0 **S** 16 inch depth interval. Thus, for example, the concentration of mercury at the soil surface in 1957 was assumed to range from 200% to 500% of the concentration averaged over the entire core.

Table 7-13: Soil Concentration Adjustment Factors

Year	Adjustment Factor (%)
1950 - 1954	100 - 400
1955 - 1958	200 - 500
1959 - 1962	50 - 300
1963 - 1966	50 - 300
1967 - 1970	40 - 200
1971 - 1974	10 - 100
1975 - 1978	5 - 100
1979 - 1982	3 - 50
1983 - 1986	1 - 50
1987 - 1990	2 - 50
1991 - 1994	1 - 30

Adjustment factor PDFs were defined as uniform distributions.

7.4.2 Soil and Sediment Concentrations– EFPC Floodplain Farm Family

It was assumed that the EFPC Floodplain Farm Family population resided at approximately EFPC Mile 10 and farmed and recreated primarily in this area. Soil samples collected during the EFPC RI were used to characterize exposures to this population group.

Soil concentrations– Direct contact with soil

Exposures through direct contact with soil (e.g., ingestion or dermal contact by farm family members) or ingestion of soil by livestock were evaluated using soil data that met the following description:

- Exposure point concentrations were characterized using samples collected between approximately EFPC Miles 9.5 and 10.5 (i.e., between EFPC RI creek transects X47500 and X51500) across the entire width of the 100-year floodplain (i.e., between EFPC RI northing N20 and southing S14).

- It was assumed that exposures were primarily to surface soils. Samples collected from the surface interval (0 – 16 in. bgs) were used to characterize exposure point concentrations. Historical concentrations for specific years were calculated using the adjustment factors described above.

A total of 151 soil samples were collected that met the above description. Concentrations in the homogenized samples were characterized by a lognormal distribution with a mean of 13 mg kg^{-1} (dry wt) and a standard deviation of 51. The maximum concentration measured was 298 mg kg^{-1} (dry wt).

Soil concentrations– Root uptake into vegetables

Exposures through uptake of mercury by vegetables grown in the floodplain were evaluated using soil data that met the following description:

- Exposure point concentrations were characterized using samples collected between approximately EFPC Miles 9.5 and 10.5 (i.e., between EFPC RI creek transects X47500 and X51500) at a distance of at least 20 meters from the creek (i.e., excluding the samples collected along the edge of the creek at EFPC RI northing N00 or southing S00), since the frequency of inundation of lower elevations of the floodplain precluded growing vegetable gardens in these areas.

- It was assumed that root uptake was associated primarily with surface soils. Therefore, samples collected from the surface interval (0 – 16 in. bgs) were used to characterize exposure point concentrations. Average concentrations of mercury in this interval were higher than in deeper intervals (i.e., 16 – 32 in. bgs and 32 – 48 in. bgs). Historical concentrations for specific years were calculated using the adjustment factors described above.

A total of 127 soil samples were collected that met the above description. Concentrations in the homogenized samples were characterized by a lognormal distribution with a mean of 7.2 mg kg^{-1} (dry wt) and a standard deviation of 20. The maximum concentration measured was 241 mg kg^{-1} (dry wt).

Sediment (in-creek) concentrations

Since limited sediment data collected in EFPC are available, EFPC Floodplain Farm Family exposures to mercury in sediment in EFPC were evaluated using soil data that met the following description:

- Exposure point concentrations for sediment were characterized using samples collected between approximately EFPC Miles 9.5 and 10.5 (i.e., between EFPC RI creek transects X47500 and X51500) on the edge of the creek (i.e., at EFPC RI northing N00 and southing S00).

- Samples collected from the surface interval (0 – 16 in. bgs) were used. Historical concentrations for specific years were calculated using the adjustment factors described above.

A total of 24 samples were collected that met this description. Concentrations in the homogenized samples were characterized by a lognormal distribution with a mean of 55 mg kg^{-1} (dry wt) and a standard deviation of 138 mg kg^{-1}. The maximum concentration was 298 mg kg^{-1} (dry wt).

7.4.3 Soil and Sediment Concentrations– Scarboro Community

It was assumed that the Scarboro Community population resided at the current location of the Scarboro Community, on the opposite side of Pine Ridge from the Y-12 Plant, and recreated primarily in this area. Soil samples collected in the Scarboro area in 1984 by ORAU were used to characterize exposures to individuals in this population via soil ingestion, dermal contact with soil, and ingestion of vegetables that took up mercury through their roots. In 1984, a total of 16 surface soil samples were collected along Hampton Road in the Scarboro Community and 41 samples were collected near the intersection of Tulsa and Tuskegee Roads. Measured mercury concentrations were low (maximum concentration 3.8 mg kg^{-1}). While EFPC does not flow through the Scarboro Community itself, the creek is close enough that children who were residents of the Scarboro Community likely visited the creek for fishing and other recreational activity, particularly since Oak Ridge was segregated until the late 1950s and community members did not have access to many of the city of Oak Ridge facilities. Exposure point concentrations for sediment were therefore characterized using soil samples collected during the EFPC RI along EFPC.

Soil concentrations-- Direct contact with soil and Root uptake into vegetables

Exposures of the Scarboro Community population to mercury in "backyard" soil through direct contact and ingestion of vegetables that took up mercury from soil were evaluated using soil data that met the following description:

- Exposure point concentrations for soil were characterized using surface soil samples collected in the Scarboro Community (on Hampton Road and the intersection of Tulsa and Tuskegee Roads) by ORAU in 1984.

A total of 57 surface soil samples (0 to 3 inch depth interval) were collected that met the above description. These samples were characterized by a lognormal distribution with a mean of 0.35 mg kg^{-1} (dry wt) and a standard deviation of 0.63 mg kg^{-1}. The highest measurement was 3.8 mg kg^{-1} (dry wt).

Sediment (in-creek) concentrations

Exposures to mercury in sediment in EFPC were evaluated using soil data that met the following description:

- Exposure point concentrations for sediment were characterized using samples collected between approximately EFPC Miles 13 and 15 (i.e., between EFPC RI creek transects N33400 and N36700) on the edge of the creek (i.e. at EFPC RI easting E00 and westing W00).

- Samples collected from the surface interval (0-16 in. bgs) were used. Historical concentrations for specific years were calculated using the adjustment factors described above.

A total of 95 samples were collected that met this description. Concentrations in the homogenized samples were characterized by a lognormal distribution with a mean of 82 mg kg^{-1} (dry wt) and a standard deviation of 250 mg kg^{-1}. The maximum concentration was 1,590 mg kg^{-1} (dry wt).

7.4.4 Soil and Sediment Concentrations– Robertsville School Children

It was assumed that members of the Robertsville School Children population lived near the EFPC floodplain and attended school at approximately EFPC Mile 12, and that they occasionally participated in recreational activities along the creek in this area. Soil samples collected during the EFPC RI between approximately EFPC Miles 11.5 and 12.5 were used to characterize exposures to this group.

Soil concentrations-- Direct contact with soil

Exposures through direct contact with soil (e.g., ingestion or dermal contact with soil) were evaluated using soil data that met the following description:

- Exposure point concentrations for soil were characterized using samples collected between approximately EFPC Miles 11.5 and 12.5 (i.e., between EFPC RI creek transects X55000 and X59000) across the entire width of the 100-year floodplain (e.g., between EFPC RI northing N14 and southing S16).

- It was assumed that exposures occurred primarily to surface soils. Samples collected from the surface interval (0 – 16 in. bgs) were used to characterize

exposure point concentrations. Historical concentrations for specific years were calculated using the adjustment factors described above.

A total of 55 soil samples were collected that met the above description. Concentrations in the homogenized samples were characterized by a lognormal distribution with a mean of 37 mg kg^{-1} (dry wt) and a standard deviation of 183 mg kg^{-1}. The maximum concentration was 207 mg kg^{-1} (dry wt).

Sediment (in-creek) concentrations

Exposures to mercury in sediment in EFPC were evaluated using soil data that met the following description:

- Exposure point concentrations for sediment were characterized using samples collected between approximately EFPC Miles 11.5 and 12.5 (i.e., between EFPC RI creek transects X55000 and X59000) on the edge of the creek (i.e., at EFPC RI northing N00 and southing S00).

- Samples collected from the surface interval (0 – 16 in. bgs) were used. Historical concentrations for specific years were calculated using the adjustment factors described above.

A total of 24 samples were collected that met this description. Concentrations in the homogenized samples were characterized by a lognormal distribution with a mean of 71 mg kg^{-1} (dry wt) and a standard deviation of 59 mg kg^{-1}. The maximum concentration was 298 mg kg^{-1} (dry wt).

PDFs used in the dose calculations to characterize mercury concentrations in soil and sediment in the EFPC floodplain are summarized in Table 7-14.

7.5 Mercury Concentrations in Fish

As discussed in Section 5.1, the earliest measurements of mercury in fish downstream from Y-12 were made in 1970 (Sanders 1970). These and later data suggest that exposures to mercury in fish may have represented a significant pathway of exposure to mercury downstream of the Y-12 Plant. The following sections describe the assumptions used to characterize historical concentrations of mercury in fish downstream from Y-12.

Table 7-14: Probability Density Functions for Characterization of Soil/ Sediment Concentrations in the EFPC Floodplain

Parameter	PDF	
	Distribution Type	**Description**
Receptor/ Pathway- Specific Soil Concentration ($C_{soil\text{-}measured}$)		
EFPC Floodplain Farm Family soil concentration– Direct contact with soil	Lognormal	Mean = 13 mg kg^{-1} (dry wt) Standard deviation = 51 mg kg^{-1}
EFPC Floodplain Farm Family soil concentration– Root uptake into vegetables	Lognormal	Mean = 7.2 mg kg^{-1} (dry wt) Standard deviation = 20 mg kg^{-1}
EFPC Floodplain Farm Family sediment concentration	Lognormal	Mean = 55 mg kg^{-1} (dry wt) Standard deviation = 138 mg kg^{-1}
Scarboro Community soil concentration– Direct contact with soil	Lognormal	Mean = 0.35 mg kg^{-1} (dry wt) Standard deviation = 0.63 mg kg^{-1}
Scarboro Community soil concentration– Root uptake into vegetables	Lognormal	Mean = 0.35 mg kg^{-1} (dry wt) Standard deviation = 0.63 mg kg^{-1}
Scarboro Community sediment concentration	Lognormal	Mean = 82 mg kg^{-1} (dry wt) Standard deviation = 250 mg kg^{-1}
Robertsville School Children soil concentration	Lognormal	Mean = 37 mg kg^{-1} (dry wt) Standard deviation = 183 mg kg^{-1}
Robertsville School Children sediment concentration	Lognormal	Mean = 71 mg kg^{-1} (dry wt) Standard deviation = 59 mg kg^{-1}

7.5.1 Approach and Assumptions

Concentrations of mercury in fish downstream from Y-12 were not measured during the period of peak releases from Y-12. Consequently, mercury concentrations in fish during this period were estimated based on:

- Concentrations of mercury in fish downstream of the ORR measured after 1970, compared to concentrations measured in sediment during the same sampling programs at approximately the same locations;
- Concentrations of mercury in fish at other sites with high mercury concentrations in sediment and/or surface water; and,
- Concentrations of mercury at depth in sediment core samples and correlations between mercury concentrations in fish vs. sediment.

The assumptions used to estimate mercury concentrations in fish are described in the following sections.

Relationships Between Methylmercury in Surface Water/ Sediment and Fish

Methylmercury (which comprises nearly 100% of the mercury in fish muscle tissue) is very bioavailable and accumulates in fish through the aquatic food web (Bloom et al. 1991)— it appears to be passed to fish primarily via their diets, with larger, longer-lived fish species at the upper end of the food web typically having the highest methylmercury concentrations in a given water body.

No data are available that describe mercury concentrations in fish near the ORR during the periods of peak mercury releases from Y-12 (the 1950s and 1960s). However, a number of investigators have demonstrated that mercury concentrations in fish can be estimated from concentrations in water and/or sediment in the nearby environment. For example, bioconcentration factors (BCFs) for accumulation of methylmercury in fish (dry weight basis), compared with the water methylmercury concentration, are on the order of 10^5 - 10^6 (USEPA Mercury Report to Congress 1997). Wren (1996) determined that mercury concentrations in fish in contaminated lakes in Canada show a strong correlation with concentrations in sediment (see Appendix R).

Historical Measurements of Mercury in Downstream Sediments

As discussed in Section 6.1, during peak release periods, minimal data are available describing mercury concentrations in water at locations downstream from Y-12 that are accessible to fishing (e.g., Poplar Creek, the Clinch River, and Watts Bar Reservoir), and no data are available describing methylmercury concentrations in surface water. However, much of the mercury released from Y-12 to EFPC was sorbed to suspended particles and sediments and eventually transported downstream and deposited in Watts Bar Reservoir, originally impounded in January 1942 (Blaylock et al. 1983; Turner et al. 1984; TVA

1985b,c,d; TVA 1986; Olsen et al 1990). A number of sediment core samples have been collected in the Clinch River and Watts Bar Reservoir. These core samples are assumed to reflect the history of mercury releases and deposition to sediment.

Sediment core samples taken from Watts Bar Reservoir in 1986 show elevated levels of mercury and cesium-137 (Cs-137) at depth in sediment. Cs-137, a radioactive isotope of cesium, was historically released from ORNL with peak releases in 1956, compared to peak releases in 1957-58 for mercury released from Y-12 (Turner et al. 1984). Both mercury and Cs-137 have a high affinity for particulate matter, suggesting that these contaminants should be highly covariant in downstream sediments (Turner et al. 1984). By assuming that the peak Cs-137 sediment concentration (at 40 to 84 cm below the sediment surface, depending on location) was deposited in 1956, the sediment layers can be "dated" and the history of deposition of Cs-137 and mercury can be documented (Olsen et al. 1990) (Table 7-15). As shown in Figure 7-7, the vertical profile of mercury concentrations in a sediment core collected at Tennessee River Mile 567.5 (the mouth of the Clinch River) corresponds well with the history of releases from Y-12 (i.e., peak sediment concentrations are reflected in the layer dated 1955-1960).

Development of Regression Equations to Describe the Relationship Between Mercury Concentrations in Sediment and Fish

The project team conducted an analysis of fish and sediment data collected near the ORR to determine if a relationship could be established between mercury concentrations in fish and sediment. A number of investigators have studied the relationship between mercury in sediment and fish in mercury-contaminated lakes in rivers in the United States and Canada (Wren 1996). Wren (1996), for example, demonstrated a strong positive correlation ($r^2 = 0.81$) between mercury concentrations in biota and in sediments from contaminated sites in Canada and the Midwestern United States (presented in Appendix R). Data from several studies spanning a number of geographical areas support this relationship (Wren 1996). Because mercury concentrations in fish can vary with site-specific factors including water chemistry, temperature, and pH, the project team examined data collected in the Oak Ridge area to determine if a relationship could be established between mercury concentrations in fish and sediment near the ORR.

Table 7-15: Vertical Distribution of Mercury and Cesium-137 in a Watts Bar Reservoir Sediment Core (Bulk Sample)Collected at Tennessee River Mile 567.5 in 1986

Sample depth (cm)	Mercury ($mg\ kg^{-1}$)	Cs-137 ($pCi\ g^{-1}$)	Historical Time Period (y) [a]
0-4	1.52	5.26	1986
4-8	1.36	5.68	1984-1985
8-12	1.77	5.06	1983
12-16	2.42	6.05	1981-1982
16-20	3.16	7.39	1980
20-24	2.32	6.51	1978-1979
24-28	2.32	6.61	1977
28-32	3.33	7.85	1975-1976
32-36	3.44	11.01	1974
36-40	2.98	9.81	1972-1973
40-48	2.91	12.03	1969-1971
48-56	4.12	13.56	1966-1968
56-64	5.1	19.02	1963-1965
64-72	6.8	22.24	1960-1962
72-80	24.4	49.08	1957-1959
80-84	19	58.36	1955-1956
84-88	6.31	28.66	1953-1954
88-92	1.24	13.01	1950-1951
92-96	0.36	18.59	1948-1949
96-100	0.35	10.64	1946-1947
100-104	0.31	13.11	---
104-108	0.34	5.84	---
108-112	0.12	0.87	---
112-116	0.09	0.25	---
116-120	0.05	0.27	---

a Assumes sediment accumulation rate of 1.4 cm/year based on peak release period in 1955 -1956

Source: Olsen et al. 1990

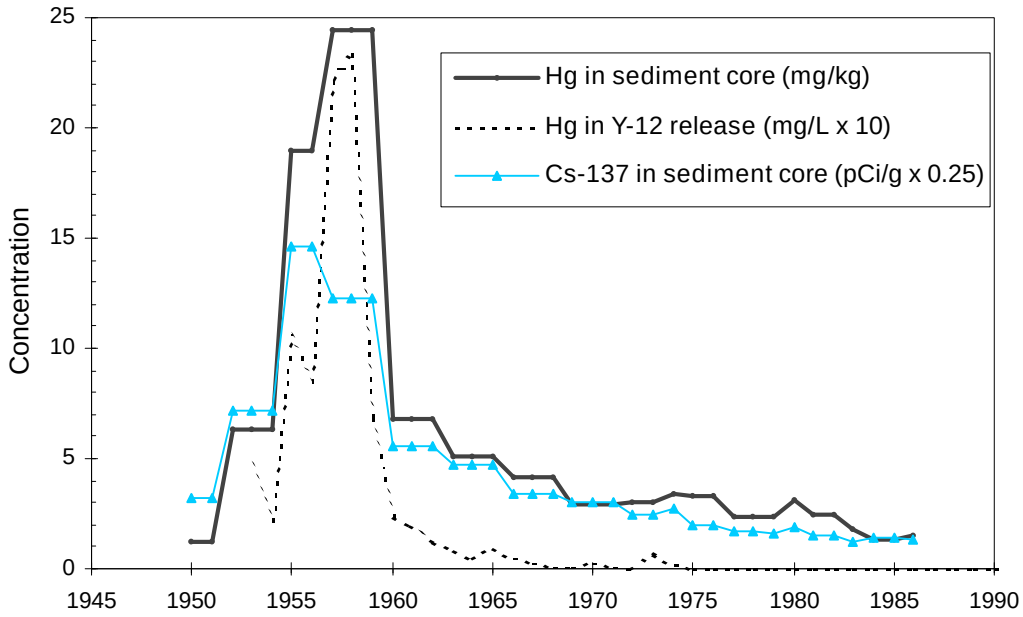


Figure 7-7: Mercury and C-137 Concentrations Measured in Dated Watts Bar Reservoir Sediment Core (TRM 567.5) vs. Y-12 Releases to EFPC

From the mid-1970s to mid-1980s, several investigators conducted large sampling programs to measure mercury concentrations in fish and sediments downstream from Y-12. These included the following:

- During May, June, and October 1976, the ORNL Environmental Sciences Division (ESD) measured mercury concentrations in fish in Poplar Creek and the Clinch River (Elwood 1984). Collection locations included Poplar Creek from the mouth of Poplar Creek to its confluence with EFPC (approximately PCM 5.5) (86 samples) and the Clinch River from CRM 4.5 to 13.5 (186 samples). ESD also measured mercury concentrations in sediment and water.

- During 1977 through 1980, the ORNL ESD measured concentrations of mercury and other contaminants in aquatic biota, including fish, in the vicinity of the ORGDP and ORNL (Loar et al. 1981a and 1981b). Sample locations included Poplar Creek (at PCM 0.5, 5.5, and 11.0) and the Clinch River (at CRM 10.5, 11.5, and 15.0). ESD also measured mercury concentrations in sediment.

- During May and July 1982, ORNL ESD measured mercury concentrations in fish in EFPC along the length of EFPC to determine whether mercury was still being released from Y-12 (Van Winkle et al. 1984). ESD also measured mercury concentrations in sediment, moss, and pasture grass.

- During 1984 and 1985, the TVA measured mercury and other contaminants in sediment, water, and aquatic biota downstream of Y-12 as part of the Instream Contaminant Study (TVA 1985a, 1985b, 1985c, 1985d, 1985e). Over 1,500 total samples were collected. Sample locations included EFPC, Poplar Creek, White Oak Creek, the Clinch River, and the Tennessee River including Watts Bar Reservoir.

Data from these sampling programs are summarized in Table 7-16. The project team evaluated mercury concentrations in fish to determine if the concentration was significantly correlated with concentrations of mercury in sediment measured during the same sampling programs at the approximate location where the fish samples were collected. If a sediment sample was not collected at the location of fish sample collection, a sediment concentration corresponding to the location of fish sample collection was interpolated using data from the nearest upstream and downstream sediment sample locations.

Several studies have shown that mercury in river and lake sediments is associated predominantly the smallest particle sizes, which have the highest specific surface areas and also frequently highest organic matter content (Elwood 1984). Cranston and Buckley (1972), for example, found that 99% of the mercury in sediments from the La Have River in Nova Scotia was associated with sediment particles <60 µm in size (Elwood 1984). During the TVA Instream Contaminant Study (TVA 1985b), mercury concentrations in sediment downstream from Y-12 were measured in several different size fractions (e.g., <62 µm, <125 µm, and <500 µm). Analysis of mercury concentrations in different size fractions at the same sampling location showed that, in general, the mercury concentration was higher in the smaller size fractions. On average, the concentration of mercury in the bulk sample was 68% of the concentration measured in the <62 µm fraction from the same sample, while the concentration in the <125 µm fraction was 91% of the concentration measured in the <62 µm fraction (TVA 1985b) .

In the 1977 through 1980 ORNL ESD studies (Loar et al. 1981a and 1981b), analyses of mercury concentrations in sediment were conducted on the bulk sample, while Van Winkle et al. (1984) measured mercury in the <125 µm fraction. For purposes of analyzing the relationship between mercury concentrations in fish and in sediment, concentrations in the <62 µm fraction were approximated by dividing bulk sample and <125 µm fraction concentrations by 0.68 and 0.91, respectively.

Table 7-16: Concentrations of Mercury in Fish Collected Downstream from the Y-12 Plant

Date	Location	Study	Year, Location, Fish Species	Minimum Hg (mg/kg, fresh)	Maximum Hg (mg/kg, fresh)	Mean Hg (mg/kg, fresh)	Mean Fish Wt (g)
May/June/Oct, 1976	Clinch R. Mile 10.5 - 11.5	Elwood, 1984	1976, Clinch R., Bigmouth buffalo	---	0.61	0.61	ND
May/June/Oct, 1976	Clinch R. Mile 12.0 (PC Mouth)	Elwood, 1984	1976, Clinch R., Bigmouth buffalo	1.68	2.08	1.88	ND
May/June/Oct, 1976	Clinch R. Mile 12.4 - 13.5	Elwood, 1984	1976, Clinch R., Bigmouth buffalo	---	0.61	0.61	ND
May/June/Oct, 1976	Clinch R. Mile 9.5 - 10.5	Elwood, 1984	1976, Clinch R., Bigmouth buffalo	0.04	0.2	0.12	ND
May/June/Oct, 1976	Clinch R. Mile 4.5 - 5.5	Elwood, 1984	1976, Clinch R., Bluegill	---	---	0.05	118
May/June/Oct, 1976	Clinch R. Mile 9.5 - 10.5	Elwood, 1984	1976, Clinch R., Bluegill	---	---	0.10	118
May/June/Oct, 1976	Clinch R. Mile 10.5 - 11.5	Elwood, 1984	1976, Clinch R., Bluegill	---	---	0.13	118
May/June/Oct, 1976	Clinch R. Mile 12.0 (PC Mouth)	Elwood, 1984	1976, Clinch R., Bluegill	---	---	0.23	118
May/June/Oct, 1976	Clinch R. Mile 12.4 - 13.5	Elwood, 1984	1976, Clinch R., Bluegill	---	---	0.10	118
May/June/Oct, 1976	Clinch R. Mile 10.5 - 11.5	Elwood, 1984	1976, Clinch R., Carp	---	0.07	0.07	ND
May/June/Oct, 1976	Clinch R. Mile 12.0 (PC Mouth)	Elwood, 1984	1976, Clinch R., Carp	0.18	0.5	0.34	ND
May/June/Oct, 1976	Clinch R. Mile 12.4 - 13.5	Elwood, 1984	1976, Clinch R., Carp	0.17	0.23	0.2	ND
May/June/Oct, 1976	Clinch R. Mile 4.5 - 5.5	Elwood, 1984	1976, Clinch R., Carp	0.14	0.26	0.15	ND
May/June/Oct, 1976	Clinch R. Mile 9.5 - 10.5	Elwood, 1984	1976, Clinch R., Carp	0.15	0.39	0.27	ND
May/June/Oct, 1976	Clinch R. Mile 4.5 - 5.5	Elwood, 1984	1976, Clinch R., Largemouth bass	---	---	0.13	210
May/June/Oct, 1976	Clinch R. Mile 9.5 - 10.5	Elwood, 1984	1976, Clinch R., Largemouth bass	---	---	0.09	210
May/June/Oct, 1976	Clinch R. Mile 10.5 - 11.5	Elwood, 1984	1976, Clinch R., Largemouth bass	---	---	0.24	210
May/June/Oct, 1976	Clinch R. Mile 12.0 (PC Mouth)	Elwood, 1984	1976, Clinch R., Largemouth bass	---	---	0.54	210
May/June/Oct, 1976	Clinch R. Mile 12.4 - 13.5	Elwood, 1984	1976, Clinch R., Largemouth bass	---	---	0.19	210
May/June/Oct, 1976	Clinch R. Mile 10.5 - 11.5	Elwood, 1984	1976, Clinch R., Sucker	0.14	0.42	0.28	ND
May/June/Oct, 1976	Clinch R. Mile 4.5 - 5.5	Elwood, 1984	1976, Clinch R., Sucker	ND	0.44	0.21	ND
May/June/Oct, 1976	Clinch R. Mile 4.5 - 5.5	Elwood, 1984	1976, Clinch R., White crappie	---	0.03	0.03	ND
May/June/Oct, 1976	Clinch R. Mile 9.5 - 10.5	Elwood, 1984	1976, Clinch R., White crappie	0.02	0.08	0.05	ND
			Clinch R. AVERAGE FOR 1976		**2.1**	**0.29**	
April, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Bluegill	0.15	0.30	0.22	31.5
April, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Gizzard shad	0.02	0.05	0.04	249
April, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., Gizzard shad	0.03	0.13	0.06	221
April/May, 1977	Clinch R. Mile 15.0	Loar et al., 1981a	1977, Clinch R., Gizzard shad	0.04	0.10	0.07	235
April, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Largemouth bass	0.04	0.15	0.08	61.4
Oct/Nov, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Largemouth bass	0.16	0.65	0.32	284
November, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., Largemouth bass	0.40	0.47	0.44	328
November, 1977	Clinch R. Mile 15.0	Loar et al., 1981a	1977, Clinch R., Largemouth bass	0.07	0.37	0.24	102
April, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Lepomis	0.05	0.28	0.16	69.4
April, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., Lepomis	0.15	0.51	0.49	84.5
Oct/Nov, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Lepomis	0.04	0.37	0.16	11.8
November, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., Lepomis	0.08	0.65	0.36	56.2
November, 1977	Clinch R. Mile 15.0	Loar et al., 1981a	1977, Clinch R., Lepomis	<0.02	1.51	0.53	34.2
Oct/Nov, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Redbreast sunfish	---	0.20	0.20	125.2
November, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., Redbreast sunfish	0.19	0.32	0.26	101
April, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., Rock bass	---	0.33	0.33	63.8
April/May, 1977	Clinch R. Mile 15.0	Loar et al., 1981a	1977, Clinch R., Sauger	---	0.29	0.29	660.2
Oct/Nov, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Sauger	0.29	0.72	0.48	693
April, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Spotted sucker	---	0.08	0.08	747
Oct/Nov, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., Striped bass	0.04	0.16	0.08	87.6
Oct/Nov, 1977	Clinch R. Mile 10.5	Loar et al., 1981a	1977, Clinch R., White bass	0.04	0.08	0.06	65.1
November, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., White bass	0.08	0.18	0.13	56.8
November, 1977	Clinch R. Mile 15.0	Loar et al., 1981a	1977, Clinch R., White bass	0.03	0.05	0.04	64.1
November, 1977	Clinch R. Mile 11.5	Loar et al., 1981a	1977, Clinch R., White crappie	---	0.33	0.33	64.3
			Clinch R. AVERAGE FOR 1977		**1.5**	**0.23**	
December, 1979	Clinch R. Mile 19.0	Loar et al., 1981b	1979, Clinch R., Bluegill	0.030	0.115	0.064	85.6
December, 1979	Clinch R. Mile 21.9	Loar et al., 1981b	1979, Clinch R., Bluegill	0.037	1.07	0.21	77.2
December, 1979	Clinch R. Mile 52 (MH Res)	Loar et al., 1981b	1979, Clinch R., Bluegill	0.031	0.077	0.061	89.7
March, 1979	Clinch R. Mile 19.0	Loar et al., 1981b	1979, Clinch R., Sauger	0.054	0.129	0.077	488
March, 1979	Clinch R. Mile 21.9	Loar et al., 1981b	1979, Clinch R., Sauger	0.063	0.197	0.103	576

Date	Location	Study	Year, Location, Fish Species	Minimum Hg (mg/kg, fresh)	Maximum Hg (mg/kg, fresh)	Mean Hg (mg/kg, fresh)	Mean Fish Wt (g)
March, 1979	Clinch R. Mile 19.0	Loar et al., 1981b	1979, Clinch R., Striped bass	0.085	0.22	0.134	1250
March, 1979	Clinch R. Mile 19.0	Loar et al., 1981b	1979, Clinch R., Yellow bass	0.076	0.148	0.10	98
			Clinch R. AVERAGE FOR 1979		**1.1**	**0.11**	
May/June 1984	Clinch R. Mile 6.0	TVA, 1985e	1984, Clinch R., Bluegill	0.12	0.33	0.19	66.2
May/June 1984	Clinch R. Mile 11.0	TVA, 1985e	1984, Clinch R., Bluegill	<0.10	0.40	0.17	92.1
May/June 1984	Clinch R. Mile 2.0	TVA, 1985e	1984, Clinch R., Bluegill	<0.10	0.13	0.065	83
May/June 1984	Clinch R. Mile 6.0	TVA, 1985e	1984, Clinch R., Largemouth bass	0.20	0.56	0.31	1350
May/June 1984	Clinch R. Mile 11.0	TVA, 1985e	1984, Clinch R., Largemouth bass	0.19	0.58	0.34	1058
May/June 1984	Clinch R. Mile 2.0	TVA, 1985e	1984, Clinch R., Largemouth bass	<0.10	0.26	0.12	660
May/June 1984	Clinch R. Mile 11.0	TVA, 1985e	1984, Clinch R., Smallmouth buffalo	<0.10	1.2	0.48	1988
			Clinch R. AVERAGE FOR 1984		**1.2**	**0.24**	
1990	Clinch R. Mile 9.5	Cook et al., 1992	1990, Clinch R., Bluegill, Channel catfish, Largemouth bass	0.186	0.77	0.43	ND
1990	Clinch R. Mile 0.5	Cook et al., 1992	1990, Clinch R., Bluegill, Channel catfish, Largemouth bass	0.044	0.22	0.11	ND
			Clinch R. AVERAGE FOR 1990		**0.77**	**0.27**	
1970	Pond/ EFPC	Sanders, 1970	1970, EFPC, Bluegill	0.41	1.3	0.76	ND
1970	EFPCM 14.2	Sanders, 1970	1970, EFPC, Carp	---	0.32	0.32	ND
1970	Pond/ EFPC	Sanders, 1970	1970, EFPC, Carp	---	0.57	0.57	ND
			EFPC AVERAGE FOR 1970		**1.3**	**0.55**	
May, 1982	EFPCM 1.3	Van Winkle et al., 1984	1982, EFPC, Bluegill, Largemouth bass, White bass	0.32	0.72	0.56	32.5
May, 1982	EFPCM 14.1	Van Winkle et al., 1984	1982, EFPC, Bluegill, Largemouth bass, White bass	0.66	2.5	1.56	61.1
May, 1982	EFPCM 14.2	Van Winkle et al., 1984	1982, EFPC, Bluegill, Largemouth bass, White bass	1.7	3.6	2.13	62.7
May, 1982	EFPCM 8.3	Van Winkle et al., 1984	1982, EFPC, Bluegill, Largemouth bass, White bass	0.73	2.2	1.39	54.6
			EFPC AVERAGE FOR 1982		**3.6**	**1.4**	
1983	Golf Course Pond (nr EFPC)	Blaylock, 1983	1983, EFPC, Bluegill	0.17	0.60	0.29	81
1983	Scarboro Pond (nr EFPC)	Blaylock, 1983	1983, EFPC, Bluegill	0.20	0.24	0.22	91.9
1983	Lower Tuskegee Crk (nr EFPC)	Blaylock, 1983	1983, EFPC, Central stoneroller	---	0.16	0.16	15
1983	Lower Tuskegee Crk (nr EFPC)	Blaylock, 1983	1983, EFPC, Creek club	0.10	0.20	0.16	38.6
1983	Scarboro Pond (nr EFPC)	Blaylock, 1983	1983, EFPC, Largemouth bass	0.28	0.74	0.42	77.3
1983	Lower Tuskegee Crk (nr EFPC)	Blaylock, 1983	1983, EFPC, Red breast sunfish	0.31	0.56	0.44	63.4
			EFPC AVERAGE FOR 1983		**0.74**	**0.28**	
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Black redhorse	---	0.57	0.57	671
May/June 1984	EFPC Mile 8.8	TVA, 1985e	1984, EFPC, Bluegill	0.51	1.0	0.80	55.9
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Bluegill	<0.10	1.2	0.75	88.5
May/June 1984	EFPC Mile 13.8	TVA, 1985e	1984, EFPC, Bluegill	---	0.54	0.54	ND
May/June 1984	EFPC Mile 13.8	TVA, 1985e	1984, EFPC, Bluegill	0.5	1.1	0.8	ND
May/June 1984	EFPC Mile 1.7	TVA, 1985e	1984, EFPC, Bluegill	0.6	0.6	0.6	ND
May/June 1984	EFPC Mile 13.8	TVA, 1985e	1984, EFPC, Carp	0.21	1.3	0.77	2193
May/June 1984	EFPC Mile 13.8	TVA, 1985e	1984, EFPC, Carp	0.2	0.2	0.2	ND
May/June 1984	EFPC Mile 1.7	TVA, 1985e	1984, EFPC, Carp	0.8	0.9	0.85	ND
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Gissard shad	---	0.12	0.12	27.2
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Green sunfish	---	0.52	0.52	31.8
May/June 1984	EFPC Mile 13.8	TVA, 1985e	1984, EFPC, Largemouth bass	0.8	1.2	1.2	294
May/June 1984	EFPC Mile 8.8	TVA, 1985e	1984, EFPC, Redbreast	0.65	1.4	0.96	71.2
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Redbreast	0.62	0.70	0.65	45.4
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Rockbass	---	1.0	1.0	118
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Warmouth	---	0.96	0.96	104
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, White sucker	0.54	1.4	0.97	376
May/June 1984	EFPC Mile 4.0	TVA, 1985e	1984, EFPC, Yellow perch	---	0.93	0.93	49.9
			EFPC AVERAGE FOR 1984		**1.4**	**0.73**	
May/June/Oct, 1976	PCM 0 - 6.0	Elwood, 1984	1976, Poplar Creek, Bigmouth buffalo	0.06	1.36	0.71	ND
May/June/Oct, 1976	PCM 0 - 6.0	Elwood, 1984	1976, Poplar Creek, Bluegill	---	---	0.40	118
May/June/Oct, 1976	PCM 0 - 6.0	Elwood, 1984	1976, Poplar Creek, Carp	0.25	0.71	0.48	ND
May/June/Oct, 1976	PCM 0 - 6.0	Elwood, 1984	1976, Poplar Creek, Largemouth bass	---	---	0.73	210
May/June/Oct, 1976	PCM 0 - 6.0	Elwood, 1984	1976, Poplar Creek, Sucker	0.13	0.41	0.27	ND

Date	Location	Study	Year, Location, Fish Species	Minimum Hg (mg/kg, fresh)	Maximum Hg (mg/kg, fresh)	Mean Hg (mg/kg, fresh)	Mean Fish Wt (g)
May/June/Oct, 1976	PCM 0 - 6.0	Elwood, 1984	1976, Poplar Creek, White crappie	0.2	0.64	0.42	ND
			POPLAR CREEK AVERAGE FOR 1976		**1.4**	**0.50**	
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, Bluegill	0.03	0.32	0.10	27
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Bluegill	0.07	0.39	0.19	42.3
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Bluegill	0.04	0.38	0.17	31
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, Channel catfish	---	0.04	0.04	39
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Channel catfish	0.08	0.44	0.24	757
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Channel catfish	0.34	0.61	0.52	926
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, Freshwater drum	---	0.15	0.15	144
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Freshwater drum	0.16	0.18	0.17	348
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, Gizzard shad	0.03	0.05	0.04	191
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Gizzard shad	0.02	0.21	0.05	275
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Gizzard shad	0.03	0.08	0.04	299
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, Largemouth bass	---	0.2	0.2	221
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Largemouth bass	0.04	0.51	0.20	74.1
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Largemouth bass	1.67	2.14	1.9	189
November, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Largemouth bass	0.55	0.87	0.71	45.1
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, Lepomis	0.02	0.06	0.04	13
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Lepomis	---	0.10	0.10	77.9
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Lepomis	0.06	0.51	0.29	28
November, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Lepomis	0.29	1.1	0.62	52.1
November, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Lepomis	0.11	0.98	0.43	53.5
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Longnose gar	0.32	0.98	0.67	2015
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Longnose gar	---	0.62	0.62	2384
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Silver redhorse	0.15	0.16	0.16	498
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, Spotted bass	0.02	0.3	0.16	5
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Spotted gar	0.30	0.52	0.41	1022
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, Spotted gar	---	0.37	0.37	1589
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Spotted sucker	0.07	0.09	0.08	409
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, Striped bass	0.08	0.21	0.14	167
April, 1977	PCM 11.0	Loar et al., 1981a	1977, Poplar Creek, White bass	0.10	0.21	0.17	410
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, White bass	0.06	0.23	0.17	370
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, White bass	0.13	0.59	0.19	492
November, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, White bass	0.04	0.30	0.16	92.1
April/May, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, White crappie	0.04	0.14	0.08	82.2
April/May, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, White crappie	0.19	0.37	0.28	111
November, 1977	PCM 0.5	Loar et al., 1981a	1977, Poplar Creek, White crappie	---	0.13	0.13	300.2
November, 1977	PCM 5.5	Loar et al., 1981a	1977, Poplar Creek, White crappie	0.29	0.81	0.66	65.4
			POPLAR CREEK AVERAGE FOR 1977		**2.1**	**0.30**	
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Blue catfish	0.06	0.07	0.065	416.5
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Blue catfish	---	0.18	0.18	1313
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, Bluegill	0.07	0.32	0.20	55.9
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Bluegill	0.33	0.69	0.44	40.1
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Bluegill	0.21	0.78	0.39	90.2
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, Channel catfish	---	1.34	1.34	1256
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Channel catfish	0.29	1.07	0.62	1100
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Channel catfish	0.11	0.12	0.12	295
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Crappie	0.31	0.63	0.44	128
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Crappie	0.11	0.48	0.28	109
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, Drum	0.07	0.08	0.075	85.7
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Drum	---	0.52	0.52	165.8
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Drum	0.08	0.30	0.18	116
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, Hybrid	---	0.28	0.28	817
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Largemouth bass	0.64	1.03	0.84	85.4

Table 7-16: Concentrations of Mercury in Fish Collected Downstream from the Y-12 Plant (Page 4 of 4)

Date	Location	Study	Year, Location, Fish Species	Minimum Hg (mg/kg, fresh)	Maximum Hg (mg/kg, fresh)	Mean Hg (mg/kg, fresh)	Mean Fish Wt (g)
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Largemouth bass	0.38	0.59	0.47	105
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Sauger	0.24	0.70	0.45	613
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Small mouth bass	---	0.58	0.58	29
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, Spotted bass	---	0.11	0.11	35.7
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, Striped bass	<0.05	0.08	0.053	88.5
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, White bass	---	<0.05	<0.05	315
1982	"PC-1"	Stiff, 1982	1982, Poplar Creek, Yellow bass	0.06	0.25	0.134	49.2
1982	"PC-2"	Stiff, 1982	1982, Poplar Creek, Yellow bass	0.07	0.52	0.29	40.5
1982	"PC-3"	Stiff, 1982	1982, Poplar Creek, Yellow catfish	0.06	0.15	0.11	606
			POPLAR CREEK AVERAGE FOR 1982		**1.3**	**0.35**	
May/June 1984	Poplar Cr. Mile 0.2	TVA, 1985e	1984, Poplar Creek, Bluegill	0.2	0.4	0.3	ND
May/June 1984	Poplar Cr. Mile 0.2	TVA, 1985e	1984, Poplar Creek, Carp	0.1	0.2	0.15	ND
May/June 1984	Poplar Cr. Mile 0.2	TVA, 1985e	1984, Poplar Creek, Channel catfish	<0.1	0.42	0.16	816
			POPLAR CREEK AVERAGE FOR 1984		**0.42**	**0.20**	
1990	Poplar Creek Mile 5.3	Cook et al., 1992	1990, Poplar Creek, Bluegill, Channel catfish, Largemouth bass	0.202	0.88	0.57	ND
1990	Poplar Creek Mile 4.6	Cook et al., 1992	1990, Poplar Creek, Bluegill, Channel catfish, Largemouth bass	0.086	0.75	0.55	ND
1990	Poplar Creek Mile 1.4	Cook et al., 1992	1990, Poplar Creek, Bluegill, Channel catfish, Largemouth bass	0.072	0.56	0.34	ND
			POPLAR CREEK AVERAGE FOR 1990		**0.88**	**0.49**	
May/June 1984	Tennessee R. Mile 572.0	TVA, 1985e	1984, Watts Bar, Bluegill	<0.10	0.17	0.062	86.2
May/June 1984	Tennessee R. Mile 558.0	TVA, 1985e	1984, Watts Bar, Bluegill	<0.10	0.18	0.078	67.6
May/June 1984	Tennessee R. Mile 572.0	TVA, 1985e	1984, Watts Bar, Largemouth bass	<0.10	0.45	0.168	1508
May/June 1984	Tennessee R. Mile 558.0	TVA, 1985e	1984, Watts Bar, Largemouth bass	<0.10	0.14	0.081	733
May/June 1984	Tennessee R. Mile 572.0	TVA, 1985e	1984, Watts Bar, Paddel fish	---	<0.10	<0.10	449
May/June 1984	Tennessee R. Mile 572.0	TVA, 1985e	1984, Watts Bar, Sauger	0.30	0.30	0.30	984
			WATTS BAR AVERAGE FOR 1984		**0.45**	**0.14**	
Dec-87	Clinch R. Mile 20.0 (Watts Bar)	TVA, 1989	1987, Watts Bar, Channel catfish	---	---	<0.10	831
			WATTS BAR AVERAGE FOR 1987			**<0.10**	
1990	Tennessee R. Mile 557.0	Cook et al., 1992	1990, Watts Bar, Bluegill, Channel catfish, Largemouth bass	0.033	0.16	0.06	ND
1990	Tennessee R. Mile 530.5	Cook et al., 1992	1990, Watts Bar, Bluegill, Channel catfish, Largemouth bass	0.032	0.25	0.10	ND
			WATTS BAR AVERAGE FOR 1990		**0.25**	**0.080**	

Note: "---" and "ND" signify not reported or not measured (I.e., no data available).

As discussed above, larger, longer-lived fish species at the upper end of the food web typically have the highest methylmercury concentrations in a given water body. For this reason, relationships between mercury in fish and in sediment were evaluated for individual fish species. Numerous investigators have shown that, within distinct species, mercury concentrations tend to increase with increasing fish size. To evaluate whether a relationship exists between fish concentration and fish weight for fish collected near the ORR, the project team evaluated data for different fish species to determine if concentrations in fish were significantly correlated with fish weight.

Linear, semi-log, and log-log regressions of mercury concentration in muscle vs. sediment concentration and fish weight were computed by species. Results of this analysis showed that fish concentrations in two resident sport species**S** bluegill and largemouth bass**S** showed reasonably good correlation with sediment concentrations. Concentrations in these fish generally decreased with increasing distance downstream from Y-12 (Figure 7-8). Although several other fish species were collected downstream of Y-12 and analyzed for mercury, the numbers of fish of these species that were collected were smaller than the numbers of bluegill or largemouth bass, and in many cases the sample size was inadequate to analyze fish concentration-fish size or fish concentration-sediment concentration relationships.

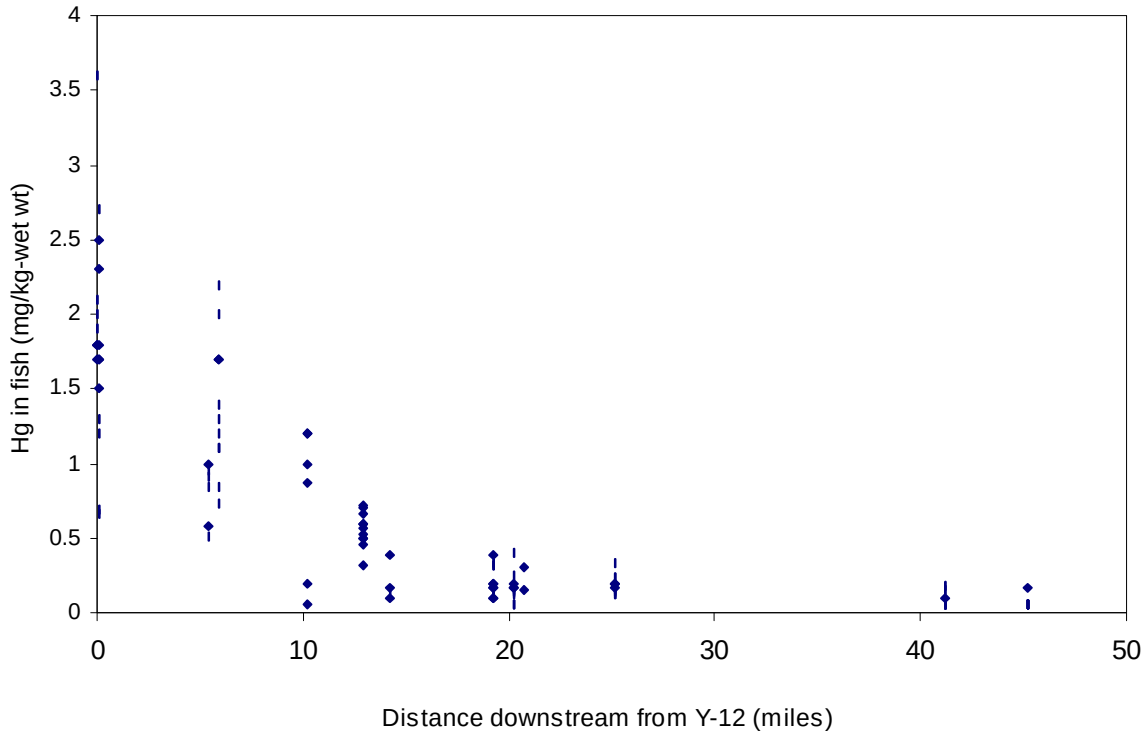


Figure 7-8: Mercury Concentrations in Bluegill vs. Downstream Distance from Y-12

Based on an analysis of the fish-sediment regression equations developed for bluegill and largemouth bass, it was determined that the linear regression resulted in the highest correlation coefficients. Multiple linear regression analysis showed a significant linear regression of mercury fish concentration vs. sediment concentration and fish weight (slope >0 for $p<0.05$) was found for bluegill, but that compared to sediment concentration, fish weight had little effect on the resulting regression equation. The concentration of mercury in largemouth bass was not correlated with fish weight.

Differences in mercury concentrations between largemouth bass and bluegill could be due to differences in size and/or feeding habits. Both species of fish are predators, but largemouth bass are primarily piscivorous (fish eaters) whereas bluegill tend to feed predominantly on benthic invertebrates (Elwood 1984). Elwood (1984) estimated that the average weights of a 2-year old largemouth bass and a 2-year old bluegill were 210 g and 118 g respectively, based on age-size relationships for fish of these species collected in reservoirs in Eastern Tennessee (Elwood 1984)**S** these are fish of a size likely to be saved by sports fishermen for consumption. Elwood (1984) showed that at the same sediment concentration, 118-g largemouth bass contain significantly greater ($p < 0.05$) concentrations of mercury than bluegill that are of the same size but older. In the current assessment, mercury concentrations were predicted for both largemouth bass and bluegill, because these fish were assumed to be representative of the size and types of fish typically caught in Eastern Tennessee waterways.

Regression equations established to describe the relationship between mercury concentrations in bluegill and largemouth bass vs. concentrations in sediment are as follows:

$$C_{fish\&bluegill} = [0.017 \times C_{sediment}] + 0.081 \qquad r^2 = 0.69 \qquad (7.17)$$

$$C_{fish\&largemouth\ bass} = [0.038 \times C_{sediment}] + 0.16 \qquad r^2 = 0.66 \qquad (7.18)$$

where C_{fish} is in mg kg^{-1} (fresh weight) and $C_{sediment}$ is in mg kg^{1} (dry weight). Because fish weight had little effect on the predicted fish concentration for bluegill, fish weight was not incorporated into the regression equation for bluegill.

These regression equations predict the mean concentration of mercury in fish for a given sediment concentration. The 5% and 95% confidence intervals around the predicted mean fish concentration associated with a given sediment concentration were also calculated using the following equation, based on the standard error of the estimated value for the dependent variable *y* (the mercury concentration in fish):

$$100\ (1\ !\ \alpha)\%\ \textit{Confidence Interval}\ '\ \text{™} \pm t_{\alpha/2}\, s \sqrt{\frac{1}{n} \% \frac{(x_p\ !\ \text{0})^2}{SS_{xx}}} \qquad (7.19)$$

where:

™	=	Predicted mean value of *y*
s	=	Estimated standard error of the regression model
x_p	=	Predicted value of *x*
0	=	Mean value of *x*
SS_{xx}	=	Sum of squares of *x*

Measurements of mercury in bluegill and largemouth bass and co-located sediment samples collected near the ORR show that, for sediment samples corresponding to the locations of collection of bluegill, concentrations in sediment ranged from 0.18 to 99 mg kg^{-1}, while for largemouth bass, concentrations in sediment ranged from 0.18 to 46 mg kg^{-1}. Consequently, it is unreasonable to apply the regression equations developed for bluegill and largemouth bass to sediment concentrations far outside of this range. Doing so would overpredict mercury concentrations in fish that correspond to these sediment concentrations, since uptake into fish is unlikely to be linearly over a wide range of sediment concentrations.

Consequently, to predict mercury concentrations in fish corresponding to high sediment and/or water concentrations (e.g., in EFPC or Poplar Creek at or around the period of peak mercury releases from Y-12), other predictors of mercury concentrations in fish were used. These included data on mercury concentrations in fish at other sites where high concentrations of mercury in water and/or sediment have been measured, and bioaccumulation factors for mercury to fish that have been developed in other studies.

7.5.2 Fish Concentration— Watts Bar Reservoir Anglers

The highest concentration of mercury measured at depth in a Watts Bar Reservoir sediment core sample was 24.4 mg kg^{-1} (dry weight, bulk sample), measured at Tennessee River Mile (TRM) 567.5 at a depth of 72 - 80 cm in 1986 (Olsen et al. 1990). Based on sediment layer dating using Cs-137 measurements, it was estimated that this peak was deposited in 1957-1959. This core was collected in the Tennessee River at the mouth of the Clinch River**S** it is expected that deposition of sediment contaminated with mercury from Y-12 in Watts Bar Reservoir would be highest at this location. Mercury concentrations measured at depth in sediment cores collected further downstream were lower (UCCND 1984a; TVA 1985). For example, the maximum mercury concentration measured in Watts Bar Reservoir cores collected during the 1984-85 TVA Instream Contaminant Study at TRMs 574.4, 552.0, 540.0, and 509.0 was 7.8 mg kg^{-1}, measured at 80 to 85 cm bgs at TRM 540.0 (total depth of core 95 cm; TVA 1985). In sediment cores collected at TRM 550 and TRM 538.3 during 1983, the maximum mercury

concentrations were 14 mg kg^{-1} and 6.9 mg kg^{-1} , measured at depths of about 95 cm bgs (total depth of core 120 cm) and 34 to 36 cm bgs (total depth of core 50 cm), respectively (Turner et al. 1984). Maximum mercury concentrations measured in sediment cores collected in different locations downstream of Y-12 are summarized in Table 7-17.

Table 7-17: Maximum Mercury Concentrations Measured in Sediment Cores Downstream of Y-12

Location of Core [a, b]	Maximum Hg Concentration (mg kg^{-1}, bulk sample, dry weight)	Depth of Maximum, cm [Total Depth of core, cm]	Reference
Poplar Creek Mile 5	460	80-84 [98]	Olsen and Cutshall 1985
Clinch River Mile 6.8	13	NA	Turner et al. 1984
Clinch River Mile 1	47	62 [87]	Turner et al. 1984
Watts Bar Reservoir (TRM 574.4)	0.30	70-76 [76]	TVA 1985
Watts Bar Reservoir (TRM 567.5)	24.4	72-80 [122]	Olsen et al. 1990
Watts Bar Reservoir (TRM 552.0)	7.5	90-95 [95]	TVA 1985
Watts Bar Reservoir (TRM 550)	14	95 [120]	Turner et al. 1984
Watts Bar Reservoir (TRM 540)	7.8	80-85 [103]	TVA 1985
Watts Bar Reservoir (TRM 538)	6.9	34-36 [50]	Turner et al. 1984

a TRM = Tennessee River Mile
b EFPC flows into Poplar Creek at Poplar Creek Mile 5, Poplar Creek flows into the Clinch River at Clinch River Mile 12, the Clinch River flows into the Tennessee River at TRM 568, and Watts Bar Dam is at TRM 530.
NA Not available

Applying the relationships between mercury concentration in sediment and fish described above to these sediment core data, historical concentrations in bluegill and largemouth bass were estimated. For each year, ranges were established for bluegill and largemouth bass using data from the Watts Bar Reservoir sediment core containing the highest mercury concentrations (TRM 567.5) and data from the Watts Bar sediment core collected downstream of the Clinch River inflow containing the lowest mercury concentrations (TRM 538). Since these sediment measurements are based on mercury concentrations in the bulk sample, sediment concentrations were divided by 0.68 to approximate the concentration in the <0.65 µm fraction used as the basis for the fish-sediment regression equations (Section 7.5.1).

It is assumed that fish swim between different areas of Watts Bar Reservoir and that anglers fish in different areas. Therefore, a range representing the annual average concentration of mercury in fish caught by an angler in a given year was derived by calculating 95th percentile confidence intervals about the predicted mean fish concentration associated with "dated" sediment concentrations in cores from TRM 567.5 and TRM 538 (Equation 7.19). Then, for each year, a triangular distribution was established using the "averaged" 5th percentile, mean, and 95th percentile as the minimum, most likely, and maximum fish concentration in the distribution.

Watts Bar Reservoir fish concentration PDFs are presented in Table 7-18. Values selected from these distributions are assumed to reflect the *mean* mercury concentration of all fish caught for consumption from Watts Bar Reservoir by an angler in a given year. Comparison of these PDFs to concentrations measured in Watts Bar Reservoir fish, shown in Table 7-16, shows that the estimated mean values are consistent with measured concentrations.

In applying the above approach to estimate concentrations in fish based on sediment core measurements, results were not constrained to be at or near zero in the early 1950s. During these years, releases from the Y-12 Plant to EFPC had begun, but were quite low compared to releases that occurred after 1952. In using the results of the dose reconstruction for fish consumption from Watts Bar Reservoir, Clinch River/Poplar Creek, and EFPC, one should keep in mind that results for the early 1950s have likely been overestimated to some degree because elevated concentrations were estimated beginning in 1950, while concentrations in fish likely built up in a more gradual manner over time. The elevated concentrations in these early years are most likely due to imprecision of the analysis and "dating" of concentrations in sediment cores.

7.5.3 Fish Concentration— Clinch River/ Poplar Creek Anglers

The maximum concentration of mercury measured in fish from Poplar Creek or the Clinch River was 2.1 mg kg^{-1}, measured in 1976 in a bigmouth buffalo from the Clinch River near the mouth of Poplar Creek (weight not recorded; Elwood 1984) (Table 7-16). Data collected by Elwood (1984) were the earliest measurements of mercury in fish from Poplar Creek or the Clinch River.

In 1985, a sediment core sample was collected by Olsen and Cutshall (1985) in Poplar Creek near the Blair Road Bridge (just downstream of the mouth of EFPC). The sample was taken to a depth of 98 cm bgs, and mercury concentrations were measured in the bulk sample at several depth intervals. Cs-137 concentrations were also measured (Table 7-19). The maximum mercury concentration was 460 mg kg^{-1} (dry weight, bulk sample) at 80 - 84 cm bgs. This concentration likely reflects maximum concentrations in Poplar Creek surface sediment during peak release years. Concentrations measured in Clinch River sediment cores further downstream were lower. The peak mercury concentrations measured in sediment cores from Clinch River Miles 6.8 and 1 were 13 mg kg^{-1} and 47 mg kg^{-1}, respectively (Turner et al. 1984, see Table 7-18). These data suggest that sediment concentrations in the Clinch River were 10- or more times lower than concentrations in Poplar Creek.

Table 7-18: Mercury Concentration Distributions Estimated for Watts Bar Reservoir Fish

Year	Min. (mg kg^{-1}, fresh)	Mean (mg kg^{-1},	Max. (mg kg^{-1}, fresh)
1950	0.005	0.13	0.24
1951	0.005	0.13	0.24
1952	0.005	0.16	0.27
1953	0.007	0.17	0.29
1954	0.022	0.19	0.32
1955	0.12	0.34	0.52
1956	0.24	0.52	0.80
1957	0.32	0.66	1.01
1958	0.37	0.74	1.14
1959	0.37	0.74	1.14
1960	0.24	0.52	0.82
1961	0.088	0.29	0.46
1962	0.088	0.29	0.46
1963	0.071	0.27	0.42
1964	0.059	0.25	0.39
1965	0.059	0.25	0.39
1966	0.052	0.23	0.37
1967	0.046	0.22	0.36
1968	0.046	0.22	0.36
1969	0.035	0.20	0.33
1970	0.029	0.19	0.32
1971	0.029	0.19	0.32
1972	0.030	0.19	0.32
1973	0.032	0.20	0.32
1974	0.036	0.21	0.34
1975	0.035	0.21	0.33
1976	0.029	0.20	0.32
1977	0.021	0.18	0.30
1978	0.021	0.18	0.30
1979	0.027	0.19	0.31
1980	0.032	0.20	0.32
1981	0.023	0.18	0.30
1982	0.017	0.17	0.29
1983	0.013	0.16	0.28
1984	0.010	0.16	0.27
1985	0.010	0.16	0.27
1986	0.010	0.16	0.27
1987	0.010	0.16	0.27
1988	0.010	0.16	0.27
1989	0.010	0.16	0.27
1990	0.010	0.16	0.27

Table 7-19: Vertical Distribution of Mercury and Cesium-137 in a Poplar Creek Sediment Core Collected Near Blair Road Bridge in 1985

Sample Depth (cm)	Mercury (mg kg^{-1})	Cs-137 (pCi g^{-1})
0-2	6.3	1.11
2-4	4.2	1.26
4-8	2.2	1.07
8-12	5.6	0.33
12-16	6.8	0.23
16-20	NA	0.18
20-24	NA	0.30
24-28	NA	0.34
28-32	NA	0.38
32-36	14	0.79
36-40	22.6	2.63
40-44	NA	1.33
44-48	NA	0.68
48-52	18	0.90
52-56	NA	1.33
56-60	NA	1.1
60-64	38.3	0.82
64-68	54.4	1.33
68-72	NA	0.87
72-76	NA	1.08
76-80	NA	1.01
80-84	460	1.06
84-88	220	1.53
88-92	40	1.71
92-96	56	4.64
96-98	NA	2.81

NA Not analyzed
Source: Olsen and Cutshall 1985

Since these sediment measurements are based on mercury concentrations in the bulk sample, sediment concentrations were divided by 0.68 to approximate the concentration in the <0.65 µm fraction used as the basis for the fish-sediment regression equations (Section 7.5.1). Applying the regression equations developed for bluegill and largemouth bass, 5th percentile, mean, and 95th percentile concentrations were estimated using the upper and lower bound sediment core data. Peak concentrations in the Poplar Creek sediment core corresponding to the period 1956 to 1961, however, significantly exceeded the range of sediment concentrations used to develop the regression equations**S** these concentrations ranged from 156 mg kg^{-1} (dry wt) to 460 mg kg^{-1} (dry wt). For these years, it is inappropriate to use the regression equations to predict mercury concentrations in fish, since the relationship is unlikely to be linear far outside of the range used to develop the equations. Consequently, peak fish concentrations for these years were estimated based on data from another highly contaminated site.

The St. Clair River and Lake St. Clair are located between Lake Huron and Lake Erie, downstream of a chloralkali plant that discharged approximately 30 lbs of elemental mercury to the St. Clair River per day in 1969. Releases were sharply curtailed beginning in 1970. Sediment concentrations of up to 1,700 mg kg^{-1} were measured in the river in 1970. Sediment concentrations in the lake were approximately 100 times lower than in the river (Wren 1996). In 1971, mercury concentrations were measured in Lake St. Clair fish. Concentrations are summarized in Table 7-20.

Table 7-20: Mercury Concentrations Measured in Fish in Lake St. Clair During 1971

Species	Weight (kg)	Mercury Concentration (mg kg^{-1}, wet wt)	
	Mean	Mean	Range
Bluegill	0.14	1.2	0.69 - 1.5
Carp	2.4	1.7	0.30 - 3.3
Channel catfish	2	1.5	0.76 - 2.8
Gizzard shad	0.31	0.16	0.04 - 0.28
Rock bass	0.21	3.1	1.4 - 6.3
Smallmouth bass	0.67	3.3	1.1 - 7.2
Muskellunge	6.6	7.1	1.8 - 23.0

As shown, the maximum mercury concentration in fish from Lake St. Clair was 23.0 mg kg^{-1} in a muskellunge. However, this is a very large fish (mean weight 6.6 kg, or about 14.5 lbs) that is near the top of the aquatic food chain. Large predatory fish generally have higher mercury concentrations than fish that are lower in the food chain, due in part to their longer life (Huckabee et al. 1979). Concentrations in fish of the size and species that are present in the Poplar Creek/ Clinch River system (i.e., bluegill, carp, catfish, shad, rock bass, smallmouth bass) were much lower (ranging from <0.05 mg kg^{-1} to 7.2 mg kg^{-1}), with mean concentrations of 3.3 mg kg^{-1} or less.

These data suggest that mean concentrations in Poplar Creek/ Clinch River fish during the years of peak releases of mercury likely did not exceed 7 mg kg^{-1}, and likely averaged 3.3 mg kg^{-1} or less. These figures were used to describe the upper-bound fish concentrations in Poplar Creek during peak release years, in lieu of using the regression equations. Concentrations in the Clinch River sediment core (the lower-bound sediment core) corresponding with these years where within the range of concentrations used to develop the regression equations; therefore, the regression equations were used to estimate lower-bound fish concentrations for these years.

A range representing the annual average concentration of mercury in fish caught by an angler in Clinch River/ Poplar Creek in a given year was derived by calculating 95th percentile confidence intervals about the predicted mean fish concentration associated with "dated" sediment concentrations in the cores from Poplar Creek and the Clinch River (Equation 7.19). Then, for each year, a triangular distribution was established using the "averaged" 5th percentile, mean, and 95th percentile as the minimum, most likely, and maximum fish concentration in the distribution.

Fish concentration ranges estimated for fish caught for consumption from Clinch River and Poplar Creek are presented in Table 7-21. Values selected from these distributions are assumed to reflect the *mean* mercury concentration of all fish caught for consumption from Clinch River and Poplar Creek by an angler in a given year. Comparison of these estimated mean values to concentrations measured in Clinch River and Poplar Creek fish, summarized in Table 7-16, shows that the estimated mean values are consistent with measured concentrations.

The earliest measurements of mercury in fish from EFPC were made in 1970 by Sanders (1970). Concentrations of mercury measured in three carp, collected near EFPC Mile 14, ranged from 0.32 to 0.76 mg kg^{-1} (fresh weight). The highest concentrations of mercury measured in EFPC fish were measured in May 1982 (Van Winkle et al. 1984); concentrations in bluegill, largemouth bass, and white bass collected between EFPC Miles 1.3 and 14.2 ranged from 0.32 to 3.6 mg kg^{-1} (fresh weight). Table 7-16 summarizes concentrations measured in EFPC fish during these and other sampling programs.

Since mercury releases from Y-12 to EFPC were highest between 1953 and 1962, it is likely that mercury concentrations in fish in EFPC were highest during these years. Because of the poor water quality in the creek during this period, however, populations of sport fish species in EFPC were likely small. Surveys of recreational fish abundance in EFPC conducted as part of the Y-12 Plant Biological Monitoring and Abatement Program (BMAP) between 1987 and 1996 identified a number of sport fish species in EFPC, including bluegill (*Lepomis macrochirus*), largemouth bass *(Micropterus salmoides*), rock bass (*Ambloplites rupestris*), white sucker (*Catostomus commersoni*), redbreast sunfish (*Lepomis auritis*), green sunfish (*Lepomis cyanellus*), warmouth (*Lepomis gulosis*), redear sunfish (*Lepomis microlophus*), and hybrid sunfish (*Lepomis*) (Barnthouse and Deppen 1996). However, these estimates of fish abundance in EFPC reflect the beneficial effects of improvements in water treatment by both the city of Oak Ridge and the Y-12 Plant during the early 1970s and early 1980s (Barnthouse and Deppen 1996). Prior to these improvements, the population of most sport species was likely substantially lower.

Table 7-21: Mercury Concentration Distributions Estimated for Poplar Creek/ Clinch River Fish

Year	Min. (mg kg^{-1}, fresh)	Mean (mg kg^{-1}, fresh)	Max. (mg kg^{-1}, fresh)
1950	0.61	1.1	1.8
1951	0.61	1.1	1.8
1952	0.61	1.1	1.8
1953	0.74	1.4	2.2
1954	0.62	1.2	1.8
1955	0.46	0.90	1.3
1956	0.88	2.2	4.3
1957	1.2	2.6	5.1
1958	1.1	2.5	4.9
1959	1.0	2.4	4.6
1960	0.90	2.2	4.4
1961	0.79	2.0	4.1
1962	0.70	1.9	3.5
1963	0.66	1.2	1.9
1964	0.51	0.97	1.5
1965	0.42	0.82	1.3
1966	0.37	0.73	1.1
1967	0.30	0.63	0.98
1968	0.24	0.52	0.81
1969	0.26	0.55	0.87
1970	0.27	0.58	0.90
1971	0.29	0.60	0.94
1972	0.30	0.62	0.97
1973	0.18	0.43	0.67
1974	0.17	0.41	0.64
1975	0.15	0.39	0.61
1976	0.14	0.37	0.57
1977	0.12	0.34	0.52
1978	0.11	0.31	0.49
1979	0.095	0.29	0.46
1980	0.082	0.27	0.43
1981	0.065	0.25	0.39
1982	0.033	0.19	0.32
1983	0.028	0.18	0.30
1984	0.066	0.25	0.39
1985	0.075	0.26	0.41
1986	0.075	0.26	0.41
1987	0.075	0.26	0.41
1988	0.075	0.26	0.41
1989	0.075	0.26	0.41
1990	0.075	0.26	0.41

7.5.4 Fish Concentration— Scarboro Community and EFPC Floodplain Farm Family

Anecdotal evidence suggests a small number of Oak Ridge residents may have caught fish in EFPC during the 1950s and 1960s (DaMassa 1995), although residents may have caught these fish in tributary streams to EFPC rather than in EFPC itself. However, since it is possible that residents caught and consumed fish from EFPC during the years of peak releases from Y-12, the project team estimated mercury concentrations in fish for these years.

Measurements of mercury at depth in stratified sediment cores collected along EFPC at the NOAA and Bruner's Study sites during the 1992 SAIC Vertical Integration Study (Table 7-12) showed that mercury concentrations at depth were as high as 3,400 mg kg^{-1} soil (dry weight) (SAIC 1994). In 1982, Van Winkle et al. (1984) measured mercury concentrations as a function of depth in a sediment core (<0.125 mm size fraction) collected in New Hope Pond. The highest concentrations (292 to 302 mg kg^{-1}) were measured at the bottom of the sediment core at depths from 70-85 cm below the surface (Table 7-22). The pond was dredged in 1973; therefore, measured concentrations represent sediment deposited since 1973. Van Winkle et al. (1984) also measured mercury concentrations in surface sediment (<0.125 mm size fraction) in EFPC in 1982 along the length of EFPC (Table 7-23). In general, mercury concentrations decreased with downstream distance from Y-12.

Data from analyses of EFPC sediment suggest that mercury concentrations in EFPC surface sediment near Y-12 are about five to 10 times higher than concentrations near the confluence of EFPC with Poplar Creek, and that surface sediment concentrations were about three times higher in 1973 than in 1982. Van Winkle et al. (1984) also observed that in 1982, mercury concentrations in bluegill caught in EFPC decreased with downstream distance from Y-12, ranging from 2.1 mg kg^{-1} (fresh weight) near Y-12 at EFPC Mile 14.2 to 0.56 mg kg^{-1} (fresh weight) at EFPC Mile 1.3.

It is difficult to estimate what mercury concentrations may have been in EFPC fish in the past because concentrations in surface water and sediment would have been much higher than levels measured after 1970, when mercury concentrations in fish were first measured. Because of the lack of data on mercury concentrations in fish downstream from Y-12 during the period of peak releases, studies of mercury concentrations in fish from other highly contaminated systems were evaluated. These studies report a maximum fish mercury concentration (in very large fish) of about 28 mg kg^{-1}. Specifically:

- The maximum muscle concentration of mercury in rainbow trout exposed to inorganic mercury ($HgCl_2$) in the laboratory was 21 mg kg^{-1} (Niimi et al. 1994). These fish were raised under laboratory rather than "environmental" conditions. Based on these and other laboratory data (Matilda et al. 1971; McKim et al. 1976), the authors suggest a body burden of 10 - 20 mg kg^{-1} mercury is lethal to fish (Niimi et al. 1994).

Table 7-22: Vertical Distribution of Mercury in a Sediment Core Collected in New Hope Pond in 1982 (Van Winkle et al. 1984)

Sample Depth (cm)	Mercury (mg kg^{-1}, <0.125 mm size fraction, dry wt)
0-5	107
5-10	108
10-15	116
15-20	110
20-25	122
25-30	174
30-35	240
35-40	220
40-45	278
45-50	**170, 166** [a]
50-55	159
55-60	**NA**
60-65	220
65-70	**NA**
70-75	302
75-80	**NA**
80-85	292
85-90	**NA**
90-95	**NA**

a Duplicate analyses
NA Not analyzed

Table 7-23: Mercury Concentrations Measured in EFPC Surface Sediment in 1982 (Van Winkle et al. 1984)

EFPC Mile	Mercury (mg kg^{-1}, <0.125 mm size fraction, dry wt)
14.2	90
14.1	62, 62 [a]
13.8	127
8.3	55
6.8	30
4.8	32
1.3	19

a Duplicate analyses

- Maximum mercury concentrations in fish from the contaminated Wabigoon-English River system in Ontario, Canada during 1970 were 24.8 mg kg^{-1} for burbot and 24.0 mg kg^{-1} for walleye from Clay Lake, and 27.8 mg kg^{-1} for northern pike from the Wabigoon River (Fimreite and Reynolds 1973; Wren 1996). An estimated 10 tons of inorganic mercury were released to this system from a chloralkali plant between 1962 and 1970 (Wren 1996).

- The maximum mercury concentration in fish from the contaminated St. Clair River/ Lake St. Clair system was 23 mg kg^{-1} for a muskellunge from Lake St. Clair in 1971 (Wren 1996). The St. Clair River and Lake St. Clair are located between Lake Huron and Lake Erie, downstream of a chloralkali plant that discharged approximately 30 lbs of elemental mercury to the St. Clair River per day in 1969. Releases were sharply curtailed beginning in 1970. Sediment concentrations of up to 1700 mg kg^{-1} were measured in the river in 1970. Sediment concentrations in the lake were approximately 100 times lower than in the river (Wren 1996).

As discussed in Section 7.5.1, large predatory fish generally have higher mercury concentrations than those lower in the food chain due in part to their longer life span (Huckabee et al. 1979). If there were fish in EFPC during the years of peak releases, it is likely that they would have been much smaller than the species discussed above (for example, the average weight of all recreational fish species in EFPC in 1987-1996 was 130 g). Mercury concentrations in these smaller fish would have been lower than in larger fish exposed to the same environmental concentrations. For example, Armstrong (1972) and Scott (1974) found that mercury concentrations were positively related to size in all fish species from Clay Lake (Huckabee et al. 1979). In Lake St. Clair in 1971 (the same year that the maximum concentration in

muskellunge of 23.0 mg kg^{-1} was measured), the concentration ranges in smaller fish of the size and species present in EFPC (i.e., bluegill, carp, catfish, shad, rock bass, smallmouth bass) ranged from <0.05 mg kg^{-1} to 7.2 mg kg^{-1}, with mean concentrations of 3.3 mg kg^{-1} or less (see Table 7-20).

For later years when sediment concentrations were likely lower, historical mercury concentrations in fish caught for consumption from EFPC were estimated based on concentrations predicted using the fish-sediment regression equations developed for bluegill and largemouth bass. For earlier years, fish concentration distributions were estimated based on data on mercury concentrations in fish measured at other sites highly contaminated with mercury, since peak sediment concentrations during these years were likely significantly higher than the sediment levels used to develop the regression equations.

It is assumed that fish swim between different areas of EFPC and that anglers fished in different areas. Therefore, a range representing the annual average concentration of mercury in fish caught by an angler in a given year was derived by calculating 95th percentile confidence intervals about the predicted mean fish concentration associated with fish caught near Y-12 and closer to the EFPC/ Poplar Creek Junction. Sediment concentrations used to estimate fish concentrations using the fish-sediment regression equations were based on data presented in Tables 7-22 and 7-23: it was assumed that surface sediment concentrations near the junction decreased between 1973 and 1982 consistent with the concentrations measured in the stratified New Hope Pond Core and that concentrations near the junction were about one-fifth of concentrations near Y-12. Sediment concentrations measured near Y-12 suggest that surface sediment concentrations at this location exceeded 100 mg kg^{-1}, and thus were outside of the range of sediment concentrations used to develop the fish-sediment regression equations. Downstream surface sediment concentrations prior to 1965 were also assumed to have exceeded the concentrations used to develop the regression equations. Therefore, for the upstream location for all years and the downstream location for 1950 to 1965, mean mercury concentrations in EFPC fish were estimated based on data collected at other sites.

Based on the data presented in Table 7-20, it was assumed that the best estimate of the *mean* mercury concentration in EFPC fish for the years for which use of the regression equations was not appropriate was 1.7 mg kg^{-1} for smaller fish (e.g., bluegill) and 3.2 mg kg^{1} for larger fish (e.g., largemouth bass). Lower bounds and upper bounds on these means were assumed to be 1 mg kg^{-1} and 4 mg kg^{-1}, and 2 mg kg^{-1} and 4.5 mg kg^{-1}, respectively. Then, for each year, a triangular distribution was established using the "averaged" 5th percentile, mean, and 95th percentile as the minimum, most likely, and maximum fish concentration in the distribution.

EFPC fish concentration PDFs are presented in Table 7-24. Values selected from these distributions are assumed to reflect the *mean* mercury concentration of all fish caught for consumption from EFPC by an angler in a given year. Comparison of these PDFs to concentrations measured in EFPC fish, shown in Table 7-16, shows that the estimated mean values are consistent with measured concentrations.

Table 7-24: Mercury Concentration Distributions Estimated for EFPC Fish

Year	Min. (mg kg^{-1}, fresh)	Mean (mg kg^{-1}, fresh)	Max. (mg kg^{-1}, fresh)
1950	1.5	2.5	4.3
1951	1.5	2.5	4.3
1952	1.5	2.5	4.3
1953	1.5	2.5	4.3
1954	1.5	2.5	4.3
1955	1.5	2.5	4.3
1956	1.5	2.5	4.3
1957	1.5	2.5	4.3
1958	1.5	2.5	4.3
1959	1.5	2.5	4.3
1960	1.5	2.5	4.3
1961	1.5	2.5	4.3
1962	1.5	2.5	4.3
1963	1.5	2.5	4.3
1964	1.5	2.5	4.3
1965	1.6	2.5	3.7
1966	1.6	2.5	3.7
1967	1.6	2.5	3.7
1968	1.6	2.4	3.7
1969	1.6	2.4	3.7
1970	1.6	2.4	3.6
1971	1.6	2.4	3.6
1972	1.5	2.3	3.6
1973	1.5	2.3	3.6
1974	1.5	2.3	3.6
1975	1.5	2.1	3.1
1976	1.4	2.0	3.0
1977	1.4	1.9	2.9
1978	1.3	1.9	2.9
1979	1.3	1.8	2.8
1980	1.2	1.7	2.7
1981	1.1	1.7	2.6
1982	1.1	1.6	2.6
1983	1.1	1.6	2.6
1984	1.0	1.4	1.9
1985	0.89	1.0	1.7
1986	0.89	1.0	1.2
1987	0.89	1.0	1.2
1988	0.89	1.0	1.2
1989	0.89	1.0	1.2
1990	0.89	1.0	1.2

This page intentionally left blank.

8.0 CHARACTERIZATION OF TRANSFER OF MERCURY TO VEGETATION AND TO MILK AND MEAT

This section describes factors derived to characterize the transfer of mercury from air and soil to vegetation and from cattle intake to milk and meat. In addition, this section describes the assumptions used to characterize the uncertainty/ variability in the transfer factors, and defines the PDFs used in the dose calculations.

Specifically, this section describes:

- **Transfer of mercury from air to vegetation**, based on estimates of mercury deposition and absorption by plants;
- **Transfer of mercury from soil to vegetation,** based on measurements of mercury concentrations in soil and plants collected in the EFPC floodplain; and
- **Transfer of mercury to milk and meat**, based on estimates of transfer of mercury ingested or inhaled by cattle.

Assumptions used by the project team to characterize transfer factor PDFs are described in the following sections.

8.1 Transfer of Mercury from Air to Above-Ground Vegetation

Direct foliar deposition and absorption of airborne mercury and mercury compounds is likely the most important source of mercury accumulation in plant leaf tissue (Beauford and Barringer 1977; de Temmerman et al. 1986; Mosbaek et al. 1988; Lindberg et al. 1995). While mercury may be taken up from the soil into the roots, translocation of the mercury from soil to the stem and upper parts of the plant is minimal (Beauford and Barringer 1977; de Temmerman et al. 1986; Mosbaek et al. 1988; Lindberg et al. 1995; Stein et al. 1996). In this assessment, it is assumed that root uptake of mercury from soil contributes to below-ground vegetation concentrations only, while uptake from air contributes to concentrations in only the above-ground portions of the plant.

The rate at which mercury is removed from the atmosphere and deposited on or absorbed by vegetation is described by the deposition velocity parameter. The principal sites of deposition of airborne mercury, predominantly comprised of elemental mercury vapor (Hg^0), in plants are probably tissues of the leaf interior, suggesting that processes controlling gas exchange at the leaf surface (e.g., stomata) and mercury assimilation at the gas-liquid interface deep within the leaf interior have a dominant role in governing deposition of Hg^0 vapor to plant canopies (Lindberg et al. 1992).

It is difficult to establish a relationship between mercury concentrations in ambient air and accumulation in plants because plants release volatile mercury to the air in addition to accumulating mercury from air and

soil (Siegel et al. 1974; Kozuchowski and Johnson 1978; Lindberg et al. 1979). The approach used in this assessment to model deposition to vegetation is described in detail in Appendix S and summarized below.

8.1.1 Model Description

The term "deposition" describes the transfer of gases or particles to surfaces exposed to the atmosphere. The rate at which mercury is removed from the atmosphere and deposited on or absorbed by vegetation, including leafy vegetables, pasture, or forest canopy, is described by the "deposition velocity" parameter. The amount of mercury deposited to the ground that is intercepted by vegetation is described by the "mass interception factor" (*r/Y*). Deposition can occur under both dry and wet conditions (e.g., during precipitation) (Equation 8.1).

$$V_{D\ (total)\&veg} = \left[V_d \times \left(\frac{r}{Y} \right)_{dry} + V_w \times \left(\frac{r}{Y} \right)_{wet} \right] \qquad (8.1)$$

where:

$V_{D\ (total)-veg}$	=	Total deposition onto vegetation ($m^3\ kg^{-1}\ d^{-1}$);
V_d	=	Total dry deposition velocity ($m\ d^{-1}$);
$(r/Y)_{dry}$	=	Mass interception factor for dry deposition onto vegetation ($m^2\ kg^{-1}$);
V_w	=	Total wet deposition velocity ($m\ d^{-1}$);
$(r/Y)_{wet}$	=	Mass interception factor for wet deposition onto vegetation ($m^2\ kg^{-1}$).

Dry deposition to vegetation can be described by the *total* dry deposition velocity (V_d) and the mass interception factor $(r/Y)_{dry}$. The *total* dry deposition velocity (V_d) relates the depositional flux of a gas or particle onto a unit area ($mg\ m^{-2}\ s^{-1}$) to the air concentration ($mg\ m^{-3}$), and is often expressed in units of centimeters per second ($cm\ s^{-1}$).

While the *total* dry deposition velocity (V_d) reflects deposition to all exposed surfaces, including vegetation, detritus, root mat, and soil, the *vegetation* dry deposition velocity (V_{d-v}) accounts for deposition to vegetation only. Many experiments that measure dry deposition velocity actually measure *vegetation* dry deposition velocity (V_{d-v}). The *vegetation* dry deposition velocity (V_{d-v}) can be mathematically described as the product of the *total* dry deposition velocity (V_d) and the interception fraction (r), which is the fraction of the net flux that is intercepted by and retained by vegetation (Equation 8.2).

$$V_{d\&v} = V_d \times r \qquad (8.2)$$

The *vegetation* dry deposition velocity ($V_{d\text{-}v}$) can be further normalized to the biomass of the vegetation (Y) [kg m^{-2}]. The *normalized* dry deposition velocity to vegetation ($V_{D\text{-}v}$) [cm^3 g^1 s^1] is mathematically given as the product of the *total* dry deposition velocity (V_d) and the mass interception factor $(r/Y)_{dry}$ (Equation 8.3).

$$V_{D\&v} = V_d \times \left(\frac{r}{Y}\right)_{dry} \tag{8.3}$$

The mass interception factor $(r/Y)_{dry}$ is specific to vegetation type (e.g., forest canopy, grasses, leafy vegetables, non-leafy vegetables).

Wet deposition describes the scavenging of a material from the atmosphere by rain or snow. The degree of wet deposition is estimated from knowledge of the washout ratio (WR) where $C_{rainwater}$ and C_{air} are the concentrations of mercury in rainwater (at ground level) and in air in the gaseous phase (Eqn. 8.4).

$$WR = \frac{C_{rainwater}\ [ng\ m^{\&3}]}{C_{air}\ [ng\ m^{\&3}]} \tag{8.4}$$

The wet deposition velocity (V_w) is given as the product of the washout ratio (WR) and the average annual precipitation rate (R), defined as the amount of rain in 365 days [m d^{-1}] (Equation 8.5).

$$V_w = WR \times R \tag{8.5}$$

The transfer of mercury carried by precipitation to vegetation is described by the mass interception factor for wet deposition $(r/Y)_{wet}$ [m^2 kg^{-1} (dry)] (Equation 8.1), defined as the fraction of the material in rain deposited per square meter of the ground surface intercepted and retained on the plant, normalized to the dry mass of the vegetation per unit area of soil.

8.1.2 Model Input Parameters

Year-round *vegetation* dry deposition velocity ($V_{d\text{-}v}$) values for mercury to a forest canopy near Oak Ridge reported by Lindberg et al. (1991) ranged from approximately 0.01 - 0.12 cm s^{-1}, with maximum values occurring in summer. For a typical growing season (i.e., May 1 - September 30), dry deposition velocities ranged from 0.04 - 0.12 cm s^{-1} (35 - 104 m d^{-1}) (Lindberg et al. 1991). This range is consistent with dry deposition velocities for mercury to forest canopy, tall grass, and alfalfa reported for other sites (range 0.03 - 0.1 cm s^{-1}) (Hildebrand et al. 1980; Barton et al. 1981; Lindberg et al. 1991; Lindberg et al. 1992; Stein et al. 1996). Based on all available data, the vegetation dry deposition velocity ($V_{d\text{-}v}$) for airborne mercury during the growing season was assumed to range from 0.03 to 0.12 cm s^{-1} (26 to 104 m d^{-1}). Although estimates by Lindberg et al. (1991) are based on deposition to a forest canopy, at heights

ranging from 21 to 43 m above the ground surface, the range is consistent with measurements of deposition to grasses and forage. Therefore, this range is considered appropriate for predicting dry deposition to ground vegetation, including fruits, vegetables, and pasture grass.

Biomass density (Y_{veg}) differs for fruits and vegetables and for pasture grass. Reported values for $Y_{leafy\ vegetables}$ range from 0.36 to 5.3 [kg (fresh wt) m^{-2}] and values for non-leafy vegetables (e.g., broccoli, cauliflower, green beans, lima beans, and sweet corn) range from 0.17 to 1.6 [kg (fresh wt) m^{-2}] (Baes and Orton 1979). The 5th and 95th percentiles of the combined data sets are approximately 0.3 and 4 [kg (fresh wt) m^{-2}], respectively. Values for Y_{forage} range from 0.04 to 1.6 [kg (dry wt) m^{-2}] (Hoffman and Baes 1979), with 5th and 95th percentiles of approximately 0.1 and 0.9 [kg (dry wt) m^{-2}]. The 5th and 95th percentile values for both vegetation types (vegetables and forage) were used to define minimum and maximum values for PDFs to describe deposition to fruits and vegetables and to pasture grass.

Limited data describing washout ratios (*WR*) for Hg^0 measured near Oak Ridge and elsewhere suggest values ranging from <500 to about 10,000 (Lindberg et al. 1994, Stein et al. 1996), with values at Oak Ridge ranging from 1,900 to 3,000. Based on these data, the washout ratio (*WR*) parameter was estimated to range from 1,000 to 10,000, with a central value of 2,500; the distribution was assumed to be triangular. Annual precipitation rates were based on measurements by the US Weather Bureau at their Oak Ridge station (near downtown Oak Ridge)(USGS 1967). Annual average precipitation at this location between 1931 and 1960 was 54.71 inches, compared to 57.85 inches at K-25 and 51.52 inches at ORNL (USGS 1967). It was therefore assumed that annual precipitation at locations of populations of interest may have varied from the concentration measured at the Oak Ridge location by ± 5%.

Mass interception factors (*r/Y*) specific to mercury were not identified. However, it was determined that mass interception factors (*r/Y*) for small aerosols, mists, and gases were most appropriate for application to mercury vapor. Mass interception factors (*r/Y*) for iodine vapor (having particle diameters typically less than 0.001 µm), 1-µm particles, 30-µm spores, and a "fine spray" of unknown particle size, deposited to grasses, range from about 1 $m^2\ kg^{-1}$ to about 4.5 $m^2\ kg^{-1}$ dry weight (Chamberlain 1970, Miller 1979b). Mass interception factors *(r/Y)* for vegetables range from about 0.2 $m^2\ kg^{-1}$ to 0.5 $m^2\ kg^{-1}$ fresh weight. The distributions were assumed to be uniform. PDFs describing the inverse biomass yield (*1/Y*) and mass interception factors (*r/Y*) (for dry and wet deposition, respectively) were assumed to be correlated (i.e., have a correlation coefficient of 1).

Mercury absorbed by plants from air or taken up through the roots from soil may be translocated to all parts of the plant (e.g., tubers, fruits, leaves, stems). However, for purposes of this assessment, it is assumed that mercury absorbed from air is reflected primarily in the mercury concentration in vegetation grown above-ground. Separate distributions were calculated for total deposition to above-ground fruits and vegetables ($Y_{(D(veg))}$) and total deposition to pasture ($Y_{(D(past))}$). Using average annual precipitation rates, total deposition to above-ground fruits and vegetables was characterized by a lognormal distribution with a mean of 124 (m d^{-1})/ ($m^2\ kg^{-1}$ (fresh wt)) and a standard deviation of 99 (m d^{-1})/ ($m^2\ kg^{-1}$(fresh wt)). Total deposition to pasture was characterized by a lognormal distribution with a mean of 416 (m d^{-1})/ ($m^2\ kg^{-1}$(dry wt)) and a standard deviation of 299 (m d^{-1})/ ($m^2\ kg^{-1}$ (dry wt)).

PDFs for characterization of mercury deposition to fruits and vegetables and to pasture grass are summarized in Table 8-1.

Table 8-1: Probability Density Functions for Characterizing Deposition of Mercury to Vegetation

Parameter		PDF	
Symbol	**Description**	**Distribution**	**Description**
V_d	Dry deposition velocity	Uniform	Minimum = 26 m d^{-1} Maximum = 104 m d^{-1}
Y_{veg}	Biomass yield for above-ground fruits and vegetables (dry deposition)	Uniform	Minimum = 0.3 kg (fresh wt) m^{-2} Maximum = 4 kg (fresh wt) m^{-2}
Y_{past}	Biomass yield for pasture (dry deposition)	Uniform	Minimum = 0.1 kg (dry wt) m^{-2} Maximum = 0.9 kg (dry wt) m^{-2}
WR	Washout ratio	Triangular	Minimum = 1,000 Central Value = 2,500 Maximum = 10,000
R	Uncertainty in annual average precipitation rate	Uniform	Minimum = -5% Maximum = +5%
r/Y_{veg}	Mass interception factor for fruits and vegetables (wet deposition)	Uniform	Minimum = 0.2 m^2 kg^{-1} (fresh wt) Maximum = 0.5 m^2 kg^{-1} (fresh wt)
r/Y_{past}	Mass interception factor for pasture (wet deposition)	Uniform	Minimum = 1 m^2 kg^{-1} (dry wt) Maximum = 4.5 m^2 kg^{-1} (dry wt)

8.2 Transfer of Mercury from Soil to Vegetation

Mercury in soil may contribute, through root uptake, to mercury concentrations in vegetation grown in the soil, although compared to uptake of airborne mercury through above-ground plant parts, uptake of mercury through the roots is minimal (Beauford and Barringer 1977; de Temmerman et al. 1986; Mosbaek et al. 1988; Lindberg et al. 1995; Stein et al. 1996). Surface contamination of vegetation by soil may also occur through wind resuspension and deposition of soil on plant surfaces, "rainsplash", or deposition of waterborne silt during flooding. Since people typically wash vegetables before consumption, the contribution of surface contamination to contaminant concentrations in vegetables is assumed to be minimal. However, the contribution of surface contamination to concentrations in pasture grass consumed by livestock may be significant.

The assumptions used to characterize the transfer of mercury from soil to vegetables and to pasture grass are described below.

8.2.1 Transfer from Soil to Below-Ground Vegetables

The B_{veg} parameter, defined as the ratio of the concentration of a contaminant in vegetation to the concentration in soil, is often used to predict concentrations in vegetables due to root uptake from soil. Most estimates of B_{veg} for mercury do not differentiate between mercury taken up from soil or from air, although it is likely that mercury in upper parts of plants accumulated from air rather than soil. In this assessment, root uptake of mercury from soil was considered to contribute to concentrations in below-ground plant parts only.

8.2.1.1 Model Description

Root uptake of mercury from soil into below-ground vegetables is described by the following ratio (Equation 8.6):

$$B_{veg} = \frac{C_{veg(b)}}{C_{soil}} \qquad (8.6)$$

where:

$C_{veg(b)}$ = Concentration of mercury in below-ground vegetables, dry weight (mg kg^{-1})

C_{soil} = Concentration of mercury in soil, dry weight (mg kg^{-1})

8.2.1.2 Model Input Parameters

B_{veg} values can vary significantly depending on a site's soil characteristics. For this reason, data collected in the Oak Ridge area only were used to characterize uptake of mercury from soil into vegetables for purposes of this assessment.

The project team identified two studies in which mercury was measured in vegetables and forage grown in the City of Oak Ridge. The first was conducted by ORAU between 1983 and 1987, and the second was conducted by SAIC as part of the EFPC Remedial Investigation (RI) in 1992. In both studies, mercury was measured in co-located soil and plant samples, and mercury B_{veg} values were calculated to establish the relationship between mercury concentrations in plants and concentrations in soil. Neither study considered the contribution of airborne mercury to plant concentrations (i.e., mercury measured in plants could have originated from air rather than soil). Data from these studies are summarized in App. T.

ORAU collected co-located plant and soil samples from locations throughout the city of Oak Ridge, including the EFPC floodplain, between 1983 and 1987 and analyzed the samples for total mercury (TDHE 1983; Hibbitts 1984; Hibbitts 1986; Gist 1987). Most of the samples were collected outside of the EFPC floodplain in areas with low to moderate soil mercury concentrations (i.e., less than 10 mg kg^{-1}). Some samples were collected from locations in the floodplain with significantly higher soil concentrations (up to 1100 mg kg^{-1}). In addition, a selection of garden vegetables was grown in a greenhouse at ORAU in various mixtures of uncontaminated and contaminated soil (from the floodplain). More than 100 sample pairs were collected, including leafy, vine, and root vegetables, forage, and pasture grass. Samples were washed prior to analysis to ensure that the data reflect mercury that is incorporated in the plant and not mercury on plant surfaces.

Concentrations measured in roots and below-ground plant parts (a total of 12 sample pairs including beets, carrots, radishes, potatoes, onions, and turnips) ranged from 0.008 to 8.3 mg kg^{-1} (dry wt). Co-located soil concentrations ranged from 0.12 to 520 mg kg^{-1} (dry wt). Below-ground plant (dry wt)/soil (dry wt) concentration ratios ($B_{veg(b)}$ values) ranged from 0.00077 to 0.26.

The second plant uptake study was conducted in the EFPC floodplain by SAIC in 1992. Sixteen co-located soil and plant sample pairs were collected from the Bruner's site (at approximately EFPC Mile 11) and analyzed for total mercury. Four below-ground vegetable samples (beets) were collected. Concentrations measured in below-ground vegetables ranged from 0.63 to 2.7 mg kg^{-1} (dry wt). Co-located soil concentrations ranged from 171 to 273 mg kg^{-1} (dry wt). Plant (dry wt)/ soil (dry wt) concentration ratios ranged from 0.0036 to 0.014.

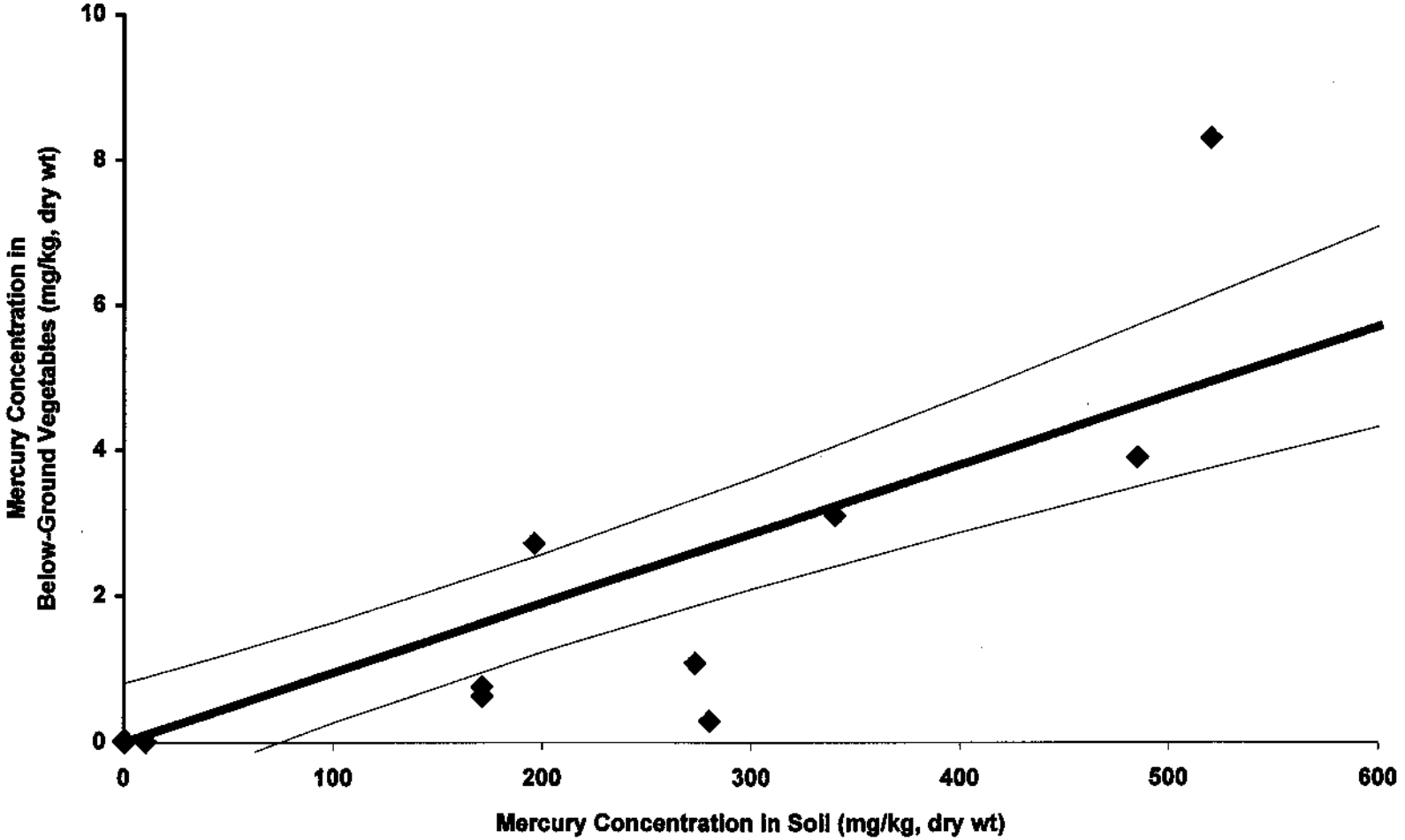
Linear regression analysis was used to find the relationship between mercury concentrations in soil and mercury concentrations in below-ground vegetables, using data from both the ORAU and SAIC studies (a total of 16 sample pairs).

The resulting regression equation is:

$$C_{veg(b)} = 0.0095 \times C_{soil} \qquad (8.7)$$

suggesting a mean $B_{veg(b)}$ value of 0.0095. The data points and regression equation are graphed in Figure 8-1 ($r^2 = 0.69$). The 95th percentile confidence interval about the predicted mean plant concentration associated with a given soil concentration was calculated using the equation described in Section 7.5.1, based on the standard error of the estimated value for the dependent variable *y* (the mercury concentration in the below-ground plant part). For soil concentrations corresponding to the central portion of the range upon which the regression equation is based (i.e., between approximately 50 mg kg^{-1} soil and 500 mg kg^{-1} soil), the 2.5th percentile and 97.5th percentile B_{veg} values are approximately 0.007 and 0.01, respectively.

Figure 8-1: Relationship between Mercury Concentration in Soil and Below-Ground Vegetables (showing the 2.5th %ile and 97.5th %ile about the predicted regression equation)



8.2.1.3 Dry-to-Wet Weight Conversion Factor for Below-Ground Vegetables

The regression equation described in Section 8.2.1.2 predicts mercury concentrations in below- ground vegetables in terms of the vegetables' dry weight (vegetation is typically dried in an oven prior to analysis). However, vegetable consumption is usually described in terms of the fresh (i.e., wet) weight of vegetables consumed per day. Therefore, predicted vegetable concentrations were converted to wet weight using a dry-to-wet weight conversion factor, based on the expected water content of different vegetables.

The ratio of dry mass to fresh mass for below-ground food crops, including beets, carrots, kohlrabi, potatoes, radishes, shallots, sweet potatoes, and turnips, ranges between 0.052 and 0.27 (arithmetic mean 0.14) (USEPA 1995a). The PDF for the dry-to-wet weight conversion factor for below-ground vegetables was assumed to be characterized by a triangular distribution, with minimum and maximum values of 0.052 and 0.27, respectively, and a central value of 0.14.

8.2.2 Transfer from Soil to Pasture

Since pasture is not washed prior to consumption by grazing livestock, surface contamination probably contributes more to contaminant intake during consumption of pasture grass than uptake of contaminants through roots (McKone 1994; Van Winkle et al. 1984; Driecer et al. 1984). However, in this assessment, ingestion of soil during grazing (including ingestion of soil on fresh pasture grass) is considered in a separate pathway. Therefore, the parameter describing transfer of mercury from soil to pasture ($B_{pasture}$) considers only mercury taken up from soil into pasture grass through roots.

Concentration ratios describing root uptake of mercury from soil by various types of forage are reported by ORAU (TDHE 1983; Hibbitts 1984; Hibbitts 1986; Gist 1987). For pasture, plant (dry weight)/soil (dry weight) concentration ratios ranging from 0.0001 to 0.002 (mean = 0.0005, n = 11) were reported. Data were insufficient, however, to derive a regression equation to describe transfer of mercury from soil to pasture. Consequently, a PDF for $B_{pasture}$ was developed assuming a logtriangular distribution with a minimum value of 0.0001, a maximum value of 0.002, and a most likely value of 0.0005.

PDFs for characterizing the transfer of mercury from soil to vegetation are summarized in Table 8-2.

Table 8-2: Probability Density Functions for Characterizing Mercury Transfer from Soil to Vegetation

Parameter		PDF	
Symbol	**Description**	**Distribution**	**Description**
$B_{veg(b)}$	Transfer from soil to below-ground vegetables	Triangular	Minimum = 0.007 Most likely = 0.0095 Maximum = 0.01
$Dw_{veg(b)}$	Dry-to-Wet weight conversion factor for below-ground vegetables	Triangular	Minimum = 0.052 Most likely = 0.14 Maximum = 0.27
$B_{pasture}$	Transfer from soil to pasture	Logtriangular	Minimum = 0.0001 Most likely = 0.0005 Maximum = 0.002

8.3 Transfer of Mercury from Cattle to Milk and Meat

Biotransfer factors were derived to predict transfer of mercury to milk and meat following ingestion of contaminated water, soil, and pasture grass by grazing livestock. These biotransfer factors are discussed below and described in detail in Appendix U.

8.3.1 Biotransfer to Milk

Transfer of mercury ingested by a cow to milk can be estimated based on experimental data describing the relationship between the concentration of mercury in milk and the cow's daily intake of mercury, as follows:

$$F_m \ (d\ L^{-1}) = \frac{Steady\text{-}state\ concentration\ of\ mercury\ in\ milk\ (mg\ L^{-1})}{Average\ daily\ intake\ of\ mercury\ (mg\ d^{-1})} \qquad (8.8)$$

F_m values are reported in the literature for both inorganic and organic mercury. Ng et al. (1979) note that organic mercury transfers to milk much more efficiently than inorganic mercury, partly due to the lipophilic nature of organic mercury. However, because methylmercury comprises a very low percentage of the mercury in EFPC floodplain soil (<0.01%) (SAIC 1994b), the F_m value used in this assessment is based on experimental data describing transfer of inorganic mercury to milk.

Limited studies are available describing biotransfer of inorganic mercury to cow's milk. Potter et al. (1972) and Mullen et al. (1975) administered soluble mercury salts (mercuric chloride and mercuric nitrate, respectively) to dairy cattle. In both studies, the mercury was incorporated into a gelatin capsule and administered to the cattle in a single dose. Data from both studies resulted in F_m values of approximately

1×10^{-5} d L^{-1} (see detailed discussion in Appendix U). For purposes of this assessment the F_m parameter is described by a loguniform distribution with minimum and maximum values of 5×10^{-6} to 5×10^{-5} d L^{-1}, respectively. This distribution was used to predict transfer of mercury from water or pasture to milk.

Speciation and bioavailability studies of mercury in EFPC floodplain soil suggest that mercury in a soil matrix is less available for absorption than mercury dissolved in water. To account for the differences in solubility between mercury compounds in EFPC floodplain soil and mercury compounds used in biotransfer studies (mercuric chloride and mercuric nitrate), the F_m value was adjusted using data on desorption of mercury from EFPC floodplain soil under conditions simulating digestion described in Section 5.2.3 (Barnett and Turner 1995). Average total soluble mercury ranged from 0.3 to 14% in 19 of 20 samples, and 46% in one sample. The average percentage of total soluble mercury was 5.3%. In contrast, 100% of mercuric chloride dissolved under these conditions. Although mercuric acetate is about ten times more soluble than mercuric chloride (solubility of mercuric acetate is 400 g L^{-1} at 20 C; ATSDR 1994), it was assumed in this assessment that the solubility of mercuric acetate under these conditions is the same as mercuric chloride (100%). Therefore, the solubility of mercury in floodplain soil relative to the solubility of mercuric acetate was assumed to be characterized by a PDF with a mean of 0.053 and a standard deviation of 0.10 (lognormal distribution). This PDF was used to adjust the F_m value used in this assessment.

8.3.2 Biotransfer to Meat

Transfer of mercury ingested by cattle to meat can be predicted based on experimental data describing the relationship between the concentration of mercury in meat and the animal's daily intake of mercury, as follows:

$$F_f \ (d\ kg^{-1}) = \frac{Steady\text{-}state\ concentration\ of\ mercury\ in\ meat\ (mg\ kg^{-1})}{Average\ daily\ intake\ of\ mercury\ (mg\ d^{-1})} \qquad (8.9)$$

The parameter F_f is used to predict the fraction of a herbivore's daily intake of mercury that is incorporated in 1 kg of muscle tissue.

Studies describing the biotransfer of inorganic mercury to meat (described in Appendix U) suggest F_f values in the 10^{-4} d kg^{-1} range. For example, data from studies by Ansari et al. (1973), Mullen et al. (1975), Potter et al. (1972), and Vreman et al. (1986) result in F_f values ranging from 1×10^{-4} to 8×10^{-4} d kg^{-1}. For purposes of this assessment, the F_f parameter is described by a loguniform distribution with minimum and maximum values of 1×10^{-4} d kg^{-1} and 9×10^{-4} d kg^{1}, respectively. This distribution was used to predict transfer of mercury from water or pasture to meat. Again, to account for the differences in solubility between the insoluble mercury compounds in EFPC floodplain soil and the mercury compound used in the uptake studies, the F_f value used to evaluate mercury biotransfer to meat following ingestion of soil was adjusted based on the assumed relative bioavailability of mercury in soil. The solubility of mercury in floodplain soil relative to the solubility of mercuric acetate was assumed to be characterized by a PDF with a mean of 0.053 and a standard deviation of 0.10 (lognormal distribution).

PDFs for characterizing the biotransfer of mercury to milk and meat are summarized in Table 8-3.

Table 8-3: Probability Density Functions for Characterizing Biotransfer of Mercury to Milk and Meat

Parameter		PDF	
Symbol	**Description**	**Distribution**	**Description**
F_m	Biotransfer to milk	Loguniform	Minimum = 5×10^{-6} d L^{-1} Maximum = 5×10^{-5} d L^{-1}
F_f	Biotransfer to meat	Loguniform	Minimum = 1×10^{-4} d kg^{-1} Maximum = 9×10^{-4} d kg^{-1}
$B_{oral\text{-}soil}$	Relative bioavailability (for adjusting between soil and water intake)	Lognormal	Arithmetic mean = 0.053 Standard deviation = 0.10

9.0 IDENTIFICATION OF PARAMETER DISTRIBUTIONS TO CHARACTERIZE EXPOSURE IN HUMANS

This section describes population-specific exposure parameters used to estimate doses of mercury to exposed individuals, in conjunction with exposure point concentrations and biotransfer factors.

9.1 Characterization of Intake by Humans

The Task 2 team developed PDFs describing population characteristics and intake rates, including body weight; exposure duration; surface area exposed to soil, sediment, and surface water; and daily intake of vapor, soil, surface water, meat, milk, vegetables and fish. The PDFs are based on information from several sources, including site-specific data, published information, and scientific judgement. These distributions describe natural inter-individual variability in such parameters as body weight and food and water ingestion rates, as well as informational uncertainty that often accompanies many "real world" situations. Some parameters may reflect both variability and uncertainty.

PDFs developed to describe population characteristics and intake rates by the population groups evaluated in this assessment are presented for each exposure pathway in Tables 9-1 through 9-6. The assumptions used to develop these distributions are described in detail in Appendix V.

9.2 Bioavailability in Humans

Intake of mercury by humans through ingestion, inhalation, or dermal contact does not necessarily mean toxic effects will occur. As discussed in Sections 2 and 5, in general, to produce a toxic effect mercury must be absorbed into the bloodstream. That is, it must be *bioavailable*. Typically, only a fraction of mercury in an environmental medium (such as soil, food, or water) is bioavailable.

The potential for adverse health effects due to exposure to a contaminant in the environment is typically evaluated by comparing the estimated dose to a toxicity benchmark value (such as a USEPA Reference Dose (RfD) or an ATSDR Minimal Risk Level (MRL)). Often, the benchmark values are based on laboratory toxicological studies in which an easily administered form of the chemical (such as a highly soluble form dissolved in water) is given to a test animal. However, since the chemical form, exposure medium, and administration route often differ between the toxicology studies and the environmental exposure, the internal dose (i.e., the dose in the bloodstream as opposed to the dose ingested or contacted) of a chemical following an environmental exposure will likely be different from that associated with the toxicology studies upon which a benchmark value is based. Adjustments may be necessary to match the exposure estimate with the toxicity value if, for example, one is based on an absorbed dose and the other is based on an intake (i.e., administered dose) (USEPA 1989a). Adjustments may also be necessary to reflect different absorption efficiencies from different exposure media (e.g., contaminants ingested with food or soil might be less completely absorbed than contaminants ingested with water) (USEPA 1989a). As discussed in Section 5.2, these adjustments are accounted for in the dose equation by the *relative bioavailability* factor.

TABLE 9-1: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE WOLF VALLEY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
General Exposure Assumptions									
Body weight (kg)- *Adult female*	$BW_{(a)}$	Lognormal	! ! !	62	! ! !	9.5	! ! !	! ! !	USEPA 1995a
Body weight (kg)- *Child (6 mo-3 yrs)*	$BW_{(c)}$	Normal	! ! !	12	! ! !	2.2	! ! !	! ! !	USEPA 1995a
Air ÷ Humans (Inhalation) [Adult (a) and Child (c)] $Dose_{air(inh)} = (C_{air} \times U_{air(a\ or\ c)} \times f_h \times [f_{ho(a\ or\ c)} \% ((1 - f_{ho(a\ or\ c)}) \times r_{io})] \times B_{inh}) / BW_{(a\ or\ c)}$									
Inhalation rate ($m^3\ d^{-1}$)- *Adult female*	$U_{air(a)}$	Lognormal	! ! !	17	! ! !	3.2	! ! !	! ! !	Layton 1993
Inhalation rate ($m^3\ d^{-1}$)-*Child (6 mo-3 yrs)*	$U_{air(c)}$	Lognormal	! ! !	5.9	! ! !	1.1	! ! !	! ! !	Layton 1993
Fraction of time at home (unitless)- *Adult female*	$f_{h(a)}$	Triangular	! ! !	! ! !	0.94	! ! !	0.68	0.98	Prof. judgement
Fraction of time at home (unitless)- *Child (6 mo- 3 yrs)*	$f_{h(c)}$	Triangular	! ! !	! ! !	0.96	! ! !	0.88	1.0	Prof. judgement
Fraction of time at home outdoors (unitless) - *Adult female*	$f_{ho(a)}$	Triangular	! ! !	! ! !	0.18	! ! !	0.064	0.28	Prof. judgement
Fraction of time at home outdoors (unitless) -*Child (6 mo-3 yrs)*	$f_{ho(c)}$	Triangular	! ! !	! ! !	0.087	! ! !	0.022	0.22	Prof. judgement
Indoor-to-outdoor ratio (unitless)	r_{io}	Uniform	! ! !	! ! !	! ! !	! ! !	0.3	0.95	
Relative bioavailability of mercury in air following inhalation (unitless)	B_{inh}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Air ÷ Above-ground Fruits and Vegetables ÷ Humans [Adult (a) and Child(c)] $Dose_{air\&veg} = (C_{air} \times V_{D(veg)} \times \left(\frac{1 - e^{-k_w\ T_{g(v)}}}{k_w}\right) \times f_w \times U_{veg\&e(a\ or\ c)} \times B_{oral\&food}) / BW_{(a\ or\ c)}$									
Total deposition onto above-ground fruits and vegetables ($m\ d^{-1}$)/($m^2\ kg^{-1}$)	$V_{D(veg)}$	Lognormal	! ! !	124	! ! !	99	! ! !	! ! !	Calculated: See Sect. 8.1

TABLE 9-1: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE WOLF VALLEY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Weathering rate for vegetables (d^{-1})	k_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for vegetables (d)	$T_{g(v)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	90	Prof. judgement
Fraction of contaminant remaining after washing (unitless)	f_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.5	1.0	IAEA 1992, 1994; Ng et al. 1978
Ingestion rate of homegrown above-ground fruits and vegetables ($kg\ d^{-1}$) - *Adult female*	$U_{veg\text{-}e(a)}$	Lognormal	! ! !	0.11	! ! !	0.12	! ! !	! ! !	USEPA 1995a
Ingestion rate of homegrown above-ground fruits and vegetables ($kg\ d^{-1}$) - *Child (6 mo-3 yrs)*	$U_{veg\text{-}e(c)}$	Lognormal	! ! !	0.056	! ! !	0.089	! ! !	! ! !	USEPA 1995a
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Air ÷ Pasture ÷ Dairy cows ÷ Milk ÷ Humans [Adult (a) and Child(c)]	$Dose_{air\&past\&milk} = (C_{air} \times V_{D(past)} \times \left(\frac{1 - e^{-k_w T_{g(p)}}}{k_w}\right) \times Q_{feed(d)} \times f_{p(d)} \times F_{m(p)} \times U_{milk(a\ or\ c)} \times B_{oral\&food} \times f_{mh}) / BW$								
Total deposition onto pasture ($m\ d^{-1}$)/($m^2\ kg^{-1}$)	$V_{D(past)}$	Lognormal	! ! !	416	! ! !	299	! ! !	! ! !	Calculated: See Sect. 8.1
Weathering rate for pasture (d^{-1})	k_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for pasture (d)	$T_{g(p)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	60	Prof. judgement
Ingestion rate of feed (dry weight) by dairy cattle ($kg\ d^{-1}$)	$Q_{feed(d)}$	Triangular	! ! !	! ! !	10	! ! !	7	14	Shor & Fields 1980; Dreicer et al. 1990
Fraction of feed consumed by dairy cattle that was pasture from the floodplain (unitless)	$f_{p(d)}$	Triangular	! ! !	! ! !	0.6	! ! !	0.4	0.75	Shor & Fields 1980; Koranda 1965
Biotransfer factor from cattle (pasture) intake to milk ($mg\ L^{-1}$)/($mg\ d^{-1}$)	$F_{m(p)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	5×10^{-6}	5×10^{-5}	Mullen et al. 1975; Neathery et al. 1974; Potter et al. 1972

TABLE 9-1: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE WOLF VALLEY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Ingestion rate of milk (L d^{-1}) - *Adult female*	$U_{milk(a)}$	Triangular	! ! !	0.28	! ! !	! ! !	0.12	0.85	USDA 1955a, 1955b, 1966
Ingestion rate of milk (L d^{-1}) -*Child (6 mo-3 yrs)*	$U_{milk(c)}$	Triangular	! ! !	0.71	! ! !	! ! !	0.24	1.2	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of milk consumed that was home-produced (unitless)	f_{mh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement
Air ÷ Pasture ÷ Livestock ÷ Beef ÷ Humans [Adult (a) and Child(c)]	$Dose_{air\&past\&beef} = (C_{air} \times V_{D(past)} \times \left(\frac{1 - e^{-k_w T_{g(p)}}}{k_w}\right) \times Q_{feed(b)} \times f_{p(b)} \times F_{f(p)} \times U_{beef(a\ or\ c)} \times B_{oral\&food} \times f_{bh}) / BW_{(a\ or\ c)}$								
Total deposition onto pasture (m d^{-1})/(m^2 kg^{-1})	$V_{D(past)}$	Lognormal	! ! !	416	! ! !	299	! ! !	! ! !	Calculated: See Sect. 8.1
Weathering rate for pasture (d^{-1})	k_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for pasture (d)	$T_{g(p)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	60	Prof. Judgement
Ingestion rate of feed (dry weight) by beef cattle (kg d^{-1})	$Q_{feed(b)}$	Triangular	! ! !	! ! !	9	! ! !	6	13	Baes et al. 1984; Mayland et al. 1977; NCRP 1985; Ng et al. 1978; Whicker & Kirchner 1987
Fraction of feed consumed by beef cattle that was pasture (unitless)	$f_{p(b)}$	Triangular	! ! !	! ! !	0.6	! ! !	0.4	0.75	Shor & Fields 1980
Biotransfer factor from cattle intake (pasture) to beef (mg kg^{-1})/(mg d^{-1})	$F_{f(p)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	1×10^{-4}	9×10^{-4}	Vreman et al. 1986
Ingestion rate of beef (kg d^{-1}) - *Adult female*	$U_{beef(a)}$	Triangular	! ! !	0.10	! ! !	! ! !	0.032	0.25	USDA 1955a, 1955b, 1966

TABLE 9-1: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE WOLF VALLEY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Ingestion rate of beef (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{beef(c)}$	Triangular	! ! !	0.039	! ! !	! ! !	0.010	0.11	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of beef consumed that was home-produced (unitless)	f_{bh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement

a Arithmetic mean
b Arithmetic standard deviation

TABLE 9-2: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE SCARBORO COMMUNITY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
General Exposure Assumptions									
Body weight (kg)- *Adult female*	$BW_{(a)}$	Lognormal	! ! !	62	! ! !	9.5	! ! !	! ! !	USEPA 1995a
Body weight (kg)- *Child (6 mo-3 yrs)*	$BW_{(c)}$	Normal	! ! !	12	! ! !	2.2	! ! !	! ! !	USEPA 1995a
Air ÷ Humans (Inhalation) [Adult (a) and Child (c)] $Dose_{air(inh)}$ ' $(C_{air} \times U_{air(a\ or\ c)} \times f_h \times [f_{ho(a\ or\ c)}$ % $((1$! $f_{ho(a\ or\ c)}) \times r_{io})] \times B_{inh})/\ BW_{(a\ or\ c)}$									
Inhalation rate ($m^3\ d^{-1}$)- *Adult female*	$U_{air(a)}$	Lognormal	! ! !	17	! ! !	3.2	! ! !	! ! !	Layton 1993
Inhalation rate ($m^3\ d^{-1}$)-*Child (6 mo-3 yrs)*	$U_{air(c)}$	Lognormal	! ! !	5.9	! ! !	1.1	! ! !	! ! !	Layton 1993
Fraction of time at home (unitless)- *Adult female*	$f_{h(a)}$	Triangular	! ! !	! ! !	0.88	! ! !	0.68	0.98	Prof. judgement
Fraction of time at home (unitless)- *Child (6 mo- 3 yrs)*	$f_{h(c)}$	Triangular	! ! !	! ! !	0.96	! ! !	0.88	1.0	Prof. judgement
Fraction of time at home outdoors (unitless) - *Adult female*	$f_{ho(a)}$	Triangular	! ! !	! ! !	0.14	! ! !	0.071	0.25	Prof. judgement
Fraction of time at home outdoors (unitless) -*Child (6 mo-3 yrs)*	$f_{ho(c)}$	Triangular	! ! !	! ! !	0.087	! ! !	0.022	0.22	Prof. judgement
Indoor-to-outdoor ratio (unitless)	r_{io}	Uniform	! ! !	! ! !	! ! !	! ! !	0.3	0.95	
Relative bioavailability of mercury in air following inhalation (unitless)	B_{inh}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement

TABLE 9-2: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE SCARBORO COMMUNITY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Air ÷ Above-ground Fruits and Vegetables ÷ Humans [Adult (a) and Child(c)]	$Dose_{air\&veg} = (C_{air} \times V_{D(veg)} \times \left(\frac{1 - e^{-k_w T_{g(v)}}}{k_w}\right) \times f_w \times U_{veg\&e(a\ or\ c)} \times B_{oral\&food}) / BW_{(a\ or\ c)}$								
Total deposition onto above-ground fruits and vegetables (m d^{-1})/(m^2 kg^{-1})	$V_{D(veg)}$	Lognormal	! ! !	124	! ! !	99	! ! !	! ! !	Calculated: See Sect. 8.1
Weathering rate for vegetables (d^{-1})	k_w	Lognormal	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for vegetables (d)	$T_{g(v)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	90	Prof. judgement
Fraction of contaminant remaining after washing (unitless)	f_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.5	1.0	IAEA 1992, 1994; Ng et al. 1978
Ingestion rate of homegrown above-ground fruits and vegetables (kg d^{-1}) - *Adult female*	$U_{veg\text{-}e(a)}$	Lognormal	! ! !	0.11	! ! !	0.12	! ! !	! ! !	USEPA 1995a
Ingestion rate of homegrown above-ground fruits and vegetables (kg d^{-1}) - *Child (6 mo-3 yrs)*	$U_{veg\text{-}e(c)}$	Lognormal	! ! !	0.056	! ! !	0.089	! ! !	! ! !	USEPA 1995a
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Soil ÷ Humans (Ingestion) [Adult (a) and Child (c)]	$Dose_{soil(ing)} = (C_{soil} \times U_{soil(a\ or\ c)} \times B_{oral\&soil} \times f_{sc(a\ or\ c)}) / BW_{(a\ or\ c)}$								
Ingestion rate of soil (kg d^{-1}) - *Adult female*	$U_{soil(a)}$	Lognormal	! ! !	0.000025	! ! !	0.000020	! ! !	! ! !	Calabrese & Stanek 1992
Ingestion rate of soil (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{soil(c)}$	Lognormal	! ! !	0.000075	! ! !	0.000060	! ! !	! ! !	Calabrese & Stanek 1992
Relative bioavailability of mercury in soil following ingestion (unitless)	$B_{oral\text{-}soil}$	Lognormal	! ! !	0.053	! ! !	0.10	! ! !	! ! !	Barnett & Turner 1995
Fraction of soil ingested that was contaminated (unitless) - *Adult female*	$f_{sc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.70	1.0	Prof. judgement

TABLE 9-2: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE SCARBORO COMMUNITY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Fraction of soil ingested that was contaminated (unitless) -*Child (6 mo-3 yrs)*	$f_{sc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.90	1.0	Prof. judgement
Soil ÷ Humans (Dermal Contact) [Adult (a) and Child (c)]	$Dose_{soil(derm)} = (C_{soil} \times SA_{soil(a\ or\ c)} \times SL_{soil} \times B_{derm\&soil} \times f_{sc(a\ or\ c)} \times 10^{-6}\ kg\ mg^{-1} \times$								
Surface area of exposed skin, dermal contact with soil ($cm^2\ d^{-1}$) - *Adult female*	$SA_{soil(a)}$	Lognormal	! ! !	3100	! ! !	300	! ! !	! ! !	USEPA 1995a
Surface area of exposed skin, dermal contact with soil– Surface area-to-body weight ratio ($cm^2\ kg^{-1}\ d^{-1}$) -*Child (6 mo-3 yrs)*	$SA_{soil(c)}$	Lognormal	! ! !	220	! ! !	33	! ! !	! ! !	USEPA 1995a
Soil loading on skin, dermal contact with soil (mg cm^{-2})	SL_{soil}	Lognormal	! ! !	0.52	! ! !	0.99	! ! !	! ! !	Finley et al. 1994
Relative bioavailability of mercury in soil upon dermal contact (unitless)	$B_{derm\text{-}soil}$	Log-triangular	! ! !	0.023	! ! !	! ! !	0.006	0.11	USEPA 1994
Fraction of soil dermally contacted that was contaminated (unitless) - *Adult female*	$f_{sc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.70	1.0	Prof. judgement
Fraction of soil dermally contacted that was contaminated (unitless) -*Child (6 mo-3 yrs)*	$f_{sc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.90	1.0	Prof. judgement
Soil ÷ Vegetables ÷ Humans [Adult (a) and Child(c)]	$Dose_{soil\&veg} = (C_{soil} \times B_{vegb} \times DW_v \times U_{veg\&b(a\ or\ c)} \times B_{oral\&food}) / BW_{(a\ or\ c)}$								
Concentration ratio for transfer of mercury from dry soil to below-ground vegetables (dry weight) (unitless)	B_{vegb}	Regression Equation	! ! !	0.0095	! ! !	! ! !	! ! !	! ! !	Gist 1987; SAIC 1993; See Sect. 8.2
Dry-to-wet weight conversion factor for vegetables (unitless)	DW_v	Triangular	! ! !	! ! !	0.14	! ! !	0.052	0.27	USEPA 1995a
Ingestion rate of homegrown below-ground vegetables ($kg\ d^{-1}$) - *Adult female*	$U_{veg\text{-}b(a)}$	Lognormal	! ! !	0.043	! ! !	0.052	! ! !	! ! !	USEPA 1995a
Ingestion rate of homegrown below-ground vegetables ($kg\ d^{-1}$) - *Child (6 mo-3 yrs)*	$U_{veg\text{-}b(c)}$	Lognormal	! ! !	0.034	! ! !	0.061	! ! !	! ! !	USEPA 1995a

TABLE 9-2: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE SCARBORO COMMUNITY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Sediment ÷ Humans (Ingestion) [Adult (a) and Child (c)]	$Dose_{sed(ing)} = (C_{sed} \times U_{sed(a\ or\ c)} \times B_{oral\&soil} \times f_{sc(a\ or\ c)} \times EF_{EF(a\ or\ c)}) / BW_{(a\ or\ c)}$								
Ingestion rate of sediment (kg d^{-1}) - *Adult female*	$U_{sed(a)}$	Lognormal	! ! !	0.000025	! ! !	0.000020	! ! !	! ! !	Finley et al. 1994
Ingestion rate of sediment (kg d^{-1}) - *Child*	$U_{sed(c)}$	Lognormal	! ! !	0.000075	! ! !	0.000060	! ! !	! ! !	Finley et al. 1994
Relative bioavailability of mercury in soil following ingestion (unitless)	$B_{oral\text{-}soil}$	Lognormal	! ! !	0.053	! ! !	0.10	! ! !	! ! !	Barnett & Turner 1995
Fraction of sediment ingested that was contaminated (unitless) - *Adult female*	$f_{sc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Fraction of sediment ingested that was contaminated (unitless) -*Child (6 mo-3 yrs)*	$f_{sc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Adult female*	$EF_{sed(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.066	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Child (6 mo-3 yrs)*	$EF_{sed(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.088	Prof. judgement
Sediment ÷ Humans (Dermal Contact) [Adult (a) and Child (c)]	$Dose_{sed(derm)} = (C_{sed} \times SA_{sed(a\ or\ c)} \times SL_{sed} \times B_{derm\&soil} \times f_{sc(a\ or\ c)} \times EF_{EF(a\ or\ c)} \times 10^{-6}\ mg\ kg^{-1} \times BW_{(c)}$ [*child*								
Surface area of exposed skin, dermal contact with soil (cm^2 d^{-1}) - *Adult female*	$SA_{soil(a)}$	Lognormal	! ! !	3100	! ! !	300	! ! !	! ! !	USEPA 1995a
Surface area of exposed skin, dermal contact with soil– Surface area-to-body weight ratio (cm^2 kg^{-1} d^{-1}) -*Child (6 mo-3 yrs)*	$SA_{soil(c)}$	Lognormal	! ! !	220	! ! !	33	! ! !	! ! !	USEPA 1995a
Sediment loading on skin (mg cm^{-2})	SL_{sed}	Lognormal	! ! !	0.52	! ! !	0.99	! ! !	! ! !	Finley et al. 1994
Relative bioavailability of mercury in sediment upon dermal contact (unitless)	$B_{derm\text{-}sed}$	Log-triangular	! ! !	0.023	! ! !	! ! !	0.006	0.11	USEPA 1994

TABLE 9-2: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE SCARBORO COMMUNITY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Fraction of sediment dermally contacted that was contaminated (unitless)	f_{sc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Adult female*	$EF_{sed(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.066	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Child (6 mo-3 yrs)*	$EF_{sed(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.088	Prof. judgement
Surface water ÷ Humans (Incidental ingestion) [Adult (a) and Child (c)]	$Dose_{water(inc\&ing)} = (C_{water} \times U_{water\&inc(a\ or\ c)} \times B_{oral\&water} \times f_{wc} \times ET_{EF(a\ or\ c)} \times EF_{EF(a\ or\ c)}) / BW_{(a\ or\ c)}$								
Incidental ingestion rate of surface water (L h^{-1}) - *Adult female*	$U_{water-inc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0	0.025	USEPA 1989
Incidental ingestion rate of surface water (L h^{-1}) - *Child (6 mo-3 yrs)*	$U_{water-inc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0	0.05	USEPA 1989
Relative bioavailability of mercury in water following ingestion (unitless)	$B_{oral-water}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Fraction of surface water ingested that was contaminated (unitless)	f_{wc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure time to surface water (h d^{-1}) - *Adult female*	$ET_{EF(a)}$	Triangular	! ! !	! ! !	0.25	! ! !	0.08	2.0	Prof. judgement
Exposure time to surface water (h d^{-1}) - *Child (6 mo-3 yrs)*	$ET_{EF(c)}$	Triangular	! ! !	! ! !	0.33	! ! !	0.08	3.0	Prof. judgement
Exposure frequency to surface water (d d^{-1}) - *Adult female*	$EF_{EF(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.066	Prof. judgement
Exposure frequency to surface water (d d^{-1}) -*Child (6 mo-3 yrs)*	$EF_{EF(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.088	Prof. judgement

TABLE 9-2: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE SCARBORO COMMUNITY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Surface water ÷ Humans (Dermal contact) [Adult (a) and Child (c)] $Dose_{water(derm)} = (C_{water} \times SA_{w(a\ or\ c)} \times PC \times f_{wc} \times ET_{EF(a\ or\ c)} \times EF_{EF(a\ or\ c)} \times 10^{-3}\ L\ cr$									
Surface area of exposed skin, dermal contact with surface water (cm^2) - *Adult female*	$SA_{w(a)}$	Lognormal	! ! !	8,000	! ! !	800	! ! !	! ! !	USEPA 1995a
Surface area of exposed skin, dermal contact with surface water– Surface area-to-body weight ratio ($cm^2\ kg^{-1}$) - *Child (6 mo-3 yrs)*	$SA_{w(c)}$	Lognormal	! ! !	400	! ! !	100	! ! !	! ! !	USEPA 1995a
Permeability constant for mercury ($cm\ h^{-1}$)	PC	Loguniform	! ! !	! ! !	! ! !	! ! !	0.001	0.01	USEPA 1994
Fraction of surface water dermally contacted that was contaminated (unitless)	f_{wc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure time to surface water ($h\ d^{-1}$) - *Adult female*	$ET_{sed(a)}$	Triangular	! ! !	! ! !	0.25	! ! !	0.08	2.0	Prof. judgement
Exposure time to surface water ($h\ d^{-1}$) - *Child (6 mo-3 yrs)*	$ET_{sed(c)}$	Triangular	! ! !	! ! !	0.33	! ! !	0.08	3.0	Prof. judgement
Exposure frequency to surface water ($d\ d^{-1}$) - *Adult female*	$EF_{sed(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.066	Prof. judgement
Exposure frequency to surface water ($d\ d^{-1}$) -*Child (6 mo-3 yrs)*	$EF_{sed(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0	0.088	Prof. judgement
Fish ÷ Humans (Ingestion) [Adult (a) and Child (c)] $Dose_{fish} = (C_{fish} \times U_{fish(a\ or\ c)} \times B_{oral\&food} \times f_{fc(a\ or\ c)}) / BW_{(a\ or\ c)}$									
Ingestion rate of fish from East Fork Poplar Creek ($kg\ d^{-1}$) - *Adult female*	$U_{fish(a)}$	Lognormal	! ! !	0.0012	! ! !	2.9	! ! !	! ! !	Ebert 1996
Ingestion rate of fish from East Fork Poplar Creek ($kg\ d^{-1}$) - *Child (6 mo-3 yrs)*	$U_{fish(c)}$	Custom	Adult fish consumption rate ($kg\ d^{-1}$) ×Child body weight (kg) × 1.3 / 70 kg						Ebert 1996
Relative bioavailability of methylmercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement

a Arithmetic mean unless otherwise specified
b Arithmetic standard deviation unless otherwise specified

TABLE 9-3: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE OAK RIDGE COMMUNITY– ROBERTSVILLE SCHOOL CHILDREN

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
General Exposure Assumptions									
Body weight (kg)- *Child (10-14 yrs)*	$BW_{(c\text{-}RS)}$	Lognormal	! ! !	46	! ! !	13	! ! !	! ! !	USEPA 1995a
Air ÷ Humans (Inhalation) [Robertsville School Child (c-RS)] $Dose_{air(inh)} = (C_{air} \times U_{air(c\&RS)} \times f_s \times [f_{so(c\&RS)} \% ((1 - f_{so(c\&RS)}) \times r_{io})] \times B_{inh}) / BW_{(c\&RS)}$									
Inhalation rate ($m^3\ d^{-1}$)- *Child (10-14 yrs)*	$U_{air(c\text{-}RS)}$	Lognormal	! ! !	16	! ! !	3.0	! ! !	! ! !	Layton 1993
Fraction of time at school (unitless)- *Child (10-14 yrs)*	$f_{s(c\text{-}RS)}$	Triangular	! ! !	! ! !	0.15	! ! !	0.13	0.18	Prof. judgement
Fraction of time at school outdoors (unitless) - *Child (10-14 yrs)*	$f_{so(c\text{-}RS)}$	Triangular	! ! !	! ! !	0.23	! ! !	0.16	0.39	Prof. judgement
Indoor-to-outdoor ratio (unitless)	r_{io}	Uniform	! ! !	! ! !	! ! !	! ! !	0.3	0.95	
Relative bioavailability of mercury in air following inhalation (unitless)	B_{inh}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Soil ÷ Humans (Ingestion) [Robertsville School Child (c-RS)] $Dose_{soil(ing)} = (C_{soil} \times U_{soil(c\&RS)} \times B_{oral\&soil} \times f_{sc(c\&RS)}) / BW_{(c\&RS)}$									
Ingestion rate of soil ($kg\ d^{-1}$) - *Child (10-14 yrs)*	$U_{soil(c\text{-}RS)}$	Lognormal	! ! !	0.000037	! ! !	0.000030	! ! !	! ! !	Calabrese & Stanek 1992
Relative bioavailability of mercury in soil following ingestion (unitless)	$B_{oral\text{-}soil}$	Lognormal	! ! !	0.053	! ! !	0.10	! ! !	! ! !	Barnett & Turner 1995
Fraction of soil ingested that was contaminated (unitless) -*Child (10-14 yrs)*	$f_{sc(c\text{-}RS)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.2	0.5	Prof. judgement

TABLE 9-3: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE OAK RIDGE COMMUNITY– ROBERTSVILLE SCHOOL CHILDREN

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Soil ÷ Humans (Dermal Contact) [Robertsville School Child (c-RS)] $Dose_{soil(derm)} = (C_{soil} \times SA_{soil(c\&RS)} \times SL_{soil} \times B_{derm\&soil} \times f_{sc(c\&RS)} \times 10^{-6}\ kg\ mg^{-1}) / BW_{(c\&RS)}$									
Surface area of exposed skin, dermal contact with soil ($cm^2\ d^{-1}$) -*Child (10-14 yrs)*	$SA_{soil(c\text{-}RS)}$	Lognormal	! ! !	3100	! ! !	400	! ! !	! ! !	USEPA 1995a
Soil loading on skin, dermal contact with soil ($mg\ cm^{-2}$)	SL_{soil}	Lognormal	! ! !	0.52	! ! !	0.99	! ! !	! ! !	Finley et al. 1994
Relative bioavailability of mercury in soil upon dermal contact (unitless)	$B_{derm\text{-}soil}$	Log-triangular	! ! !	0.023	! ! !	! ! !	0.006	0.11	USEPA 1994
Fraction of soil dermally contacted that was contaminated (unitless) -*Child (10-14 yrs)*	$f_{sc(c\text{-}RS)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.2	0.5	Prof. judgement
Sediment ÷ Humans (Ingestion) [Robertsville School Child (c-RS)] $Dose_{sed(ing)} = (C_{sed} \times U_{sed(c\&RS)} \times B_{oral\&soil} \times f_{sc(c\&RS)} \times EF_{EF(c\&RS)}) / BW_{(c\&RS)}$									
Ingestion rate of sediment ($kg\ d^{-1}$) - *Child (10-14 yrs)*	$U_{sed(c\text{-}RS)}$	Lognormal	! ! !	0.000038	! ! !	0.000030	! ! !	! ! !	Finley et al. 1994
Bioavailability of mercury in soil following ingestion (unitless)	$B_{oral\text{-}soil}$	Lognormal	! ! !	0.053	! ! !	0.10	! ! !	! ! !	Barnett & Turner 1995
Fraction of sediment ingested that was contaminated (unitless) - *Child (10-14 yrs)*	$f_{sc(c\text{-}RS)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure frequency to sediment ($d\ d^{-1}$) - *Child (10-14 yrs)*	$EF_{EF(c\text{-}RS)}$	Triangular	! ! !	! ! !	0.011	! ! !	0	0.050	Prof. judgement
Sediment ÷ Humans (Dermal Contact) [Robertsville School Child (c-RS)] $Dose_{sed(derm)} = (C_{sed} \times SA_{sed(c\&RS)} \times SL_{sed} \times B_{derm\&soil} \times f_{sc(c\&RS)} \times EF_{EF(c\&RS)} \times 10^{-6}\ mg\ kg^{-1}) / BW_{(c\&RS)}$									
Surface area of exposed skin, dermal contact with soil–Surface area-to-body weight ratio ($cm^2\ d^{-1}$) - *Child (10-14 yrs)*	$SA_{soil(c\text{-}RS)}$	Lognormal	! ! !	3100	! ! !	400	! ! !	! ! !	USEPA 1995a
Sediment loading on skin ($mg\ cm^{-2}$)	SL_{sed}	Lognormal	! ! !	0.52	! ! !	0.99	! ! !	! ! !	Finley et al. 1994

TABLE 9-3: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE OAK RIDGE COMMUNITY– ROBERTSVILLE SCHOOL CHILDREN

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Relative bioavailability of mercury in sediment upon dermal contact (unitless)	$B_{derm\text{-}sed}$	Log-triangular	! ! !	0.023	! ! !	! ! !	0.006	0.11	USEPA 1994
Fraction of sediment dermally contacted that was contaminated (unitless)	f_{sc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Child (10-14 yrs)*	$EF_{EF(cRS)}$	Triangular	! ! !	! ! !	0.011	! ! !	0	0.050	Prof. judgement
Surface water ÷ Humans (Incidental ingestion) [Robertsville School Child (c-RS)]	$Dose_{water(inc\&ing)} = (C_{water} \times U_{water\&inc(c\&RS)} \times B_{oral\&water} \times f_{wc} \times ET_{EF(c\&RS)} \times EF_{EF(c\&RS)}) / BW_{(c\&RS)}$								
Incidental ingestion rate of surface water (L h^{-1}) - *Child (10-14 yrs)*	$U_{water\text{-}inc(c\text{-}RS)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0	0.025	USEPA 1989
Relative bioavailability of mercury in water following ingestion (unitless)	$B_{oral\text{-}water}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Fraction of surface water ingested that was contaminated (unitless)	f_{wc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure time to surface water (h d^{-1}) - *Child (10-14 yrs)*	$ET_{EF(c\text{-}RS)}$	Triangular	! ! !	! ! !	0.5	! ! !	0.08	6.0	Prof. judgement
Exposure frequency to surface water (d d^{-1}) -*Child (10-14 yrs)*	$EF_{EF(c\text{-}RS)}$	Triangular	! ! !	! ! !	0.011	! ! !	0	0.050	Prof. judgement
Surface water ÷ Humans (Dermal contact) [Robertsville School Child (c-RS)]	$Dose_{water(derm)} = (C_{water} \times SA_{w(c\&RS)} \times PC \times f_{wc} \times ET_{EF(c\&RS)} \times EF_{EF(c\&RS)} \times 10^{\&3}\ L\ cm^{\&3}) / BW_{(c\&RS)}$								
Surface area of exposed skin, dermal contact with surface water (cm^2) - *Child (10-14 yrs)*	$SA_{w(c\text{-}RS)}$	Lognormal	! ! !	7800	! ! !	1100	! ! !	! ! !	USEPA 1995a
Permeability constant for mercury (cm h^{-1})	PC	Loguniform	! ! !	! ! !	! ! !	! ! !	0.001	0.01	USEPA 1994
Fraction of surface water dermally contacted that was contaminated (unitless)	f_{wc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement

TABLE 9-3: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE OAK RIDGE COMMUNITY– ROBERTSVILLE SCHOOL CHILDREN

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Exposure time to surface water (h d^{-1}) - *Child (10-14 yrs)*	$ET_{EF(c\text{-}RS)}$	Triangular	! ! !	! ! !	0.5	! ! !	0.08	6.0	Prof. judgement
Exposure frequency to surface water (d d^{-1}) - *Child (10-14 yrs)*	$EF_{EF(c\text{-}RS)}$	Triangular	! ! !	! ! !	0.011	! ! !	0	0.050	Prof. judgement

a Arithmetic mean unless otherwise specified
b Arithmetic standard deviation unless otherwise specified

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
General Exposure Assumptions									
Body weight (kg)- *Adult female*	$BW_{(a)}$	Lognormal	! ! !	62	! ! !	9.5	! ! !	! ! !	USEPA 1995a
Body weight (kg)- *Child (6 mo-3 yrs)*	$BW_{(c)}$	Normal	! ! !	12	! ! !	2.2	! ! !	! ! !	USEPA 1995a
Air ÷ Humans (Inhalation) [Adult (a) and Child (c)]	$Dose_{air(inh)} = (C_{air} \times U_{air(a\ or\ c)} \times f_h \times [f_{ho(a\ or\ c)} \% ((1 - f_{ho(a\ or\ c)}) \times r_{io})] \times B_{inh}) / BW_{(a\ or\ c)}$								
Inhalation rate ($m^3\ d^{-1}$)- *Adult female*	$U_{air(a)}$	Lognormal	! ! !	17	! ! !	3.2	! ! !	! ! !	Layton 1993
Inhalation rate ($m^3\ d^{-1}$)-*Child (6 mo-3 yrs)*	$U_{air(c)}$	Lognormal	! ! !	5.9	! ! !	1.1	! ! !	! ! !	Layton 1993
Fraction of time at home (unitless)- *Adult female*	$f_{h(a)}$	Triangular	! ! !	! ! !	0.94	! ! !	0.68	0.98	Prof. judgement
Fraction of time at home (unitless)- *Child (6 mo- 3 yrs)*	$f_{h(c)}$	Triangular	! ! !	! ! !	0.96	! ! !	0.88	1.0	Prof. judgement
Fraction of time at home outdoors (unitless) - *Adult female*	$f_{ho(a)}$	Triangular	! ! !	! ! !	0.18	! ! !	0.064	0.28	Prof. judgement
Fraction of time at home outdoors (unitless) -*Child (6 mo-3 yrs)*	$f_{ho(c)}$	Triangular	! ! !	! ! !	0.087	! ! !	0.022	0.22	Prof. judgement
Indoor-to-outdoor ratio (unitless)	r_{io}	Uniform	! ! !	! ! !	! ! !	! ! !	0.3	0.95	
Relative bioavailability of mercury in air following inhalation (unitless)	B_{inh}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Air ÷ Above-ground Fruits and Vegetables ÷ Humans [Adult (a) and Child(c)]	$Dose_{air\&veg} = (C_{air} \times V_{D(veg)} \times \left(\frac{1 - e^{-k_w T_{g(v)}}}{k_w}\right) \times f_w \times U_{veg\&e(a\ or\ c)} \times B_{oral\&food} / BW_{(a\ or\ c)}$								
Total deposition onto above-ground fruits and vegetables ($m\ d^{-1}$)/($m^2\ kg^{-1}$)	$V_{D(veg)}$	Lognormal	! ! !	124	! ! !	99	! ! !	! ! !	Calculated: See Sect. 8.1

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Weathering rate for vegetables (d^{-1})	k_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for vegetables (d)	$T_{g(v)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	90	Prof. judgement
Fraction of contaminant remaining after washing (unitless)	f_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.5	1.0	IAEA 1992, 1994; Ng et al. 1978
Ingestion rate of homegrown above-ground fruits and vegetables (kg d^{-1}) - *Adult female*	$U_{veg\text{-}e(a)}$	Lognormal	! ! !	0.11	! ! !	0.12	! ! !	! ! !	USEPA 1995a
Ingestion rate of homegrown above-ground fruits and vegetables (kg d^{-1}) - *Child (6 mo-3 yrs)*	$U_{veg\text{-}e(c)}$	Lognormal	! ! !	0.056	! ! !	0.089	! ! !	! ! !	USEPA 1995a
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Air ÷ Pasture ÷ Dairy cows ÷ Milk ÷ Humans [Adult (a) and Child(c)]	$Dose_{air\&past\&milk} = (C_{air} \times V_{D(past)} \times \left(\frac{1 - e^{-k_w T_{g(p)}}}{k_w}\right) \times Q_{feed(d)} \times f_{p(d)} \times F_{m(p)} \times U_{milk(a\ or\ c)} \times B_{oral\&food} \times f_{mh}) / BW$								
Total deposition onto pasture (m d^{-1})/(m^2 kg^{-1})	$V_{D(past)}$	Lognormal	! ! !	416	! ! !	299	! ! !	! ! !	Calculated: See Sect. 8.1
Weathering rate for pasture (d^{-1})	k_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for pasture (d)	$T_{g(p)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	60	Prof. judgement
Ingestion rate of feed (dry weight) by dairy cattle (kg d^{-1})	$Q_{feed(d)}$	Triangular	! ! !	! ! !	10	! ! !	7	14	Shor & Fields 1980; Dreicer et al. 1990
Fraction of feed consumed by dairy cattle that was pasture (unitless)	$f_{p(d)}$	Triangular	! ! !	! ! !	0.6	! ! !	0.4	0.75	Shor & Fields 1980; Koranda 1965
Biotransfer factor from cattle (pasture) intake to milk (mg L^{-1})/(mg d^{-1})	$F_{m(p)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	5×10^{-6}	5×10^{-5}	Mullen et al. 1975; Neathery et al. 1974; Potter et al. 1972

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Ingestion rate of milk (L d^{-1}) - *Adult female*	$U_{milk(a)}$	Triangular	! ! !	0.28	! ! !	! ! !	0.12	0.85	USDA 1955a, 1955b, 1966
Ingestion rate of milk (L d^{-1}) -*Child (6 mo-3 yrs)*	$U_{milk(c)}$	Triangular	! ! !	0.71	! ! !	! ! !	0.24	1.2	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of milk consumed that was home-produced (unitless)	f_{mh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement
Air ÷ Pasture ÷ Livestock ÷ Beef ÷ Humans [Adult (a) and Child(c)]	$Dose_{air\&past\&beef} = (C_{air} \times V_{D(past)} \times \left(\frac{1 - e^{-k_w T_{g(p)}}}{k_w}\right) \times Q_{feed(b)} \times f_{p(b)} \times F_{f(p)} \times U_{beef(a\ or\ c)} \times B_{oral\&food} \times f_{bh}) / BW_{(a\ or\ c)}$								
Total deposition onto pasture (m d^{-1})/(m^2 kg^{-1})	$V_{D(past)}$	Lognormal	! ! !	416	! ! !	299	! ! !	! ! !	Calculated: See Sect. 8.1
Weathering rate for pasture (d^{-1})	k_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for pasture (d)	$T_{g(p)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	60	Prof. Judgement
Ingestion rate of feed (dry weight) by beef cattle (kg d^{-1})	$Q_{feed(b)}$	Triangular	! ! !	! ! !	9	! ! !	6	13	Baes et al. 1984; Mayland et al. 1977; NCRP 1985; Ng et al. 1978; Whicker & Kirchner 1987
Fraction of feed consumed by beef cattle that was pasture (unitless)	$f_{p(b)}$	Triangular	! ! !	! ! !	0.6	! ! !	0.4	0.75	Shor & Fields 1980
Biotransfer factor from cattle intake (pasture) to beef (mg kg^{-1})/(mg d^{-1})	$F_{f(p)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	1×10^{-4}	9×10^{-4}	Vreman et al. 1986
Ingestion rate of beef (kg d^{-1}) - *Adult female*	$U_{beef(a)}$	Triangular	! ! !	0.10	! ! !	! ! !	0.032	0.25	USDA 1955a, 1955b, 1966
Ingestion rate of beef (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{beef(c)}$	Triangular	! ! !	0.039	! ! !	! ! !	0.010	0.11	USDA 1955a, 1955b, 1966

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of beef consumed that was home-produced (unitless)	f_{bh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement
Soil ÷ Humans (Ingestion) [Adult (a) and Child (c)]	$Dose_{soil(ing)} = (C_{soil} \times U_{soil(a\ or\ c)} \times B_{oral\&soil} \times f_{sc(a\ or\ c)}) / BW_{(a\ or\ c)}$								
Ingestion rate of soil (kg d^{-1}) - *Adult female*	$U_{soil(a)}$	Lognormal	! ! !	0.000025	! ! !	0.000020	! ! !	! ! !	Calabrese & Stanek 1992
Ingestion rate of soil (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{soil(c)}$	Lognormal	! ! !	0.000075	! ! !	0.000060	! ! !	! ! !	Calabrese & Stanek 1992
Relative bioavailability of mercury in soil following ingestion (unitless)	$B_{oral\text{-}soil}$	Lognormal	! ! !	0.053	! ! !	0.10	! ! !	! ! !	Barnett & Turner 1995
Fraction of soil ingested that was contaminated (unitless) - *Adult female*	$f_{sc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.70	1.0	Prof. judgement
Fraction of soil ingested that was contaminated (unitless) -*Child (6 mo-3 yrs)*	$f_{sc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.90	1.0	Prof. judgement
Soil ÷ Humans (Dermal Contact) [Adult (a) and Child (c)]	$Dose_{soil(derm)} = (C_{soil} \times SA_{soil(a\ or\ c)} \times SL_{soil} \times B_{derm\&soil} \times f_{sc(a\ or\ c)} \times 10^{-6}\ kg\ mg^{-1} \times$								
Surface area of exposed skin, dermal contact with soil ($cm^2\ d^{-1}$) - *Adult female*	$SA_{soil(a)}$	Lognormal	! ! !	3100	! ! !	300	! ! !	! ! !	USEPA 1995a
Surface area of exposed skin, dermal contact with soil–Surface area-to-body weight ratio ($cm^2\ kg^{-1}\ d^{-1}$) -*Child (6 mo-3 yrs)*	$SA_{soil(c)}$	Lognormal	! ! !	220	! ! !	33	! ! !	! ! !	USEPA 1995a
Soil loading on skin, dermal contact with soil (mg cm^{-2})	SL_{soil}	Lognormal	! ! !	0.52	! ! !	0.99	! ! !	! ! !	Finley et al. 1994
Relative bioavailability of mercury in soil upon dermal contact (unitless)	$B_{derm\text{-}soil}$	Log-triangular	! ! !	0.023	! ! !	! ! !	0.006	0.11	USEPA 1994
Fraction of soil dermally contacted that was contaminated (unitless) - *Adult female*	$f_{sc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.70	1.0	Prof. judgement

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Fraction of soil dermally contacted that was contaminated (unitless) -*Child (6 mo-3 yrs)*	$f_{sc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.90	1.0	Prof. judgement
Soil ÷ Vegetables ÷ Humans [Adult (a) and Child(c)]	$Dose_{soil\&veg} = (C_{soil} \times B_{vegb} \times DW_v \times U_{veg\&b(a\ or\ c)} \times B_{oral\&food}) / BW_{(a\ or\ c)}$								
Concentration ratio for transfer of mercury from dry soil to below-ground vegetables (dry weight) (unitless)	B_{vegb}	Regression Equation	! ! !	0.0095	! ! !	! ! !	! ! !	! ! !	Gist 1987; SAIC 1993; See Sect. 8.2
Dry-to-wet weight conversion factor for vegetables (unitless)	DW_v	Triangular	! ! !	! ! !	0.14	! ! !	0.052	0.27	USEPA 1995a
Ingestion rate of homegrown below-ground vegetables (kg d^{-1}) - *Adult female*	$U_{veg\text{-}b(a)}$	Lognormal	! ! !	0.043	! ! !	0.052	! ! !	! ! !	USEPA 1995a
Ingestion rate of homegrown below-ground vegetables (kg d^{-1}) - *Child (6 mo-3 yrs)*	$U_{veg\text{-}b(c)}$	Lognormal	! ! !	0.034	! ! !	0.061	! ! !	! ! !	USEPA 1995a
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Soil ÷ Dairy cows ÷ Milk ÷ Humans [Adult (a) and Child(c)]	$Dose_{soil\&milk} = (C_{soil} \times Q_{soil(d)} \times f_{sdc} \times F_{m(s)} \times U_{milk(a\ or\ c)} \times B_{oral\&food} \times f_{mh}) / BW_{(a\ or\ c)}$								
Ingestion rate of soil by dairy cattle (kg d^{-1})	$Q_{soil(d)}$	Uniform/ Custom	Minimum = 0.04 × $f_{p(d)}$ × $Q_{feed(d)}$; Maximum = 0.08 × $f_{p(d)}$ × $Q_{feed(d)}$						Healy 1968
Ingestion rate of feed (dry weight) by dairy cattle (kg d^{-1})	$Q_{feed(d)}$	Triangular	! ! !	! ! !	10	! ! !	7	14	Shor & Fields 1980; Dreicer et al. 1990
Fraction of feed consumed by dairy cattle that was pasture (unitless)	$f_{p(d)}$	Triangular	! ! !	! ! !	0.6	! ! !	0.4	0.75	Shor & Fields 1980; Koranda 1965
Fraction of soil ingested by dairy cattle that was contaminated (unitless)	f_{sdc}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Biotransfer from cattle intake (soil) to milk (mg L^{-1}) / (mg d^{-1})	$F_{m(s)}$	Custom	Bioavailability of mercury following ingestion of soil ($B_{oral\text{-}soil}$) × Biotransfer from cattle intake (water) to beef ($F_{f(w)}$)						
Ingestion rate of milk (L d^{-1}) - *Adult female*	$U_{milk(a)}$	Triangular	! ! !	0.28	! ! !	! ! !	0.12	0.85	USDA 1955a, 1955b, 1966

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Ingestion rate of milk (L d^{-1}) -*Child (6 mo-3 yrs)*	$U_{milk(c)}$	Triangular	! ! !	0.71	! ! !	! ! !	0.24	1.2	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of milk consumed that was home-produced (unitless)	f_{mh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b; Prof. judgement
Soil ÷ Livestock ÷ Beef ÷ Humans [Adult (a) and Child(c)]	$Dose_{soil\&beef} = (C_{soil} \times Q_{soil(b)} \times f_{sbc} \times F_{f(s)} \times U_{beef(a\ or\ c)} \times B_{oral\&food} \times f_{bh}) / BW_{(a\ or\ c)}$								
Ingestion rate of soil by beef cattle (kg d^{-1})	$Q_{soil(b)}$	Uniform/ Custom	Minimum = 0.04 × $f_{p(b)}$ × $Q_{feed(b)}$; Maximum = 0.08 × $f_{p(b)}$ × $Q_{feed(b)}$						Healy 1968
Ingestion rate of feed (dry weight) by beef cattle (kg d^{-1})	$Q_{feed(b)}$	Triangular	! ! !	! ! !	9	! ! !	6	13	Baes et al. 1984; Mayland et al. 1977; NCRP 1985; Ng et al. 1978; Whicker & Kirchner 1987
Fraction of feed consumed by beef cattle that was pasture (unitless)	$f_{p(b)}$	Triangular	! ! !	! ! !	0.6	! ! !	0.4	0.75	Shor & Fields 1980
Fraction of soil ingested by beef cattle that was contaminated (unitless)	f_{sbc}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Biotransfer from cattle intake (soil) to beef (mg kg^{-1}) / (mg d^{-1})	$F_{f(s)}$	Custom	Bioavailability of mercury following ingestion of soil ($B_{oral\text{-}soil}$) × Biotransfer from cattle intake (water) to beef ($F_{f(w)}$)						
Ingestion rate of beef (kg d^{-1}) - *Adult female*	$U_{beef(a)}$	Triangular	! ! !	0.10	! ! !	! ! !	0.032	0.25	USDA 1955a, 1955b, 1966
Ingestion rate of beef (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{beef(c)}$	Triangular	! ! !	0.039	! ! !	! ! !	0.010	0.11	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of beef consumed that was home-produced (unitless)	f_{bh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Soil ÷ Pasture ÷ Dairy cows ÷ Milk ÷ Humans [Adult (a) and Child(c)]	$Dose_{soil\&past\&milk} = (C_{soil} \times B_{past} \times Q_{feed(d)} \times f_{p(d)} \times F_{m(p)} \times U_{milk(a\ or\ c)} \times B_{oral\&food} \times f_{mh}) / BW_{(a\ or\ c)}$								
Concentration ratio for transfer of mercury from dry soil to pasture (dry weight) (unitless)	B_{past}	Log-triangular	! ! !	! ! !	0.0005	! ! !	0.0001	0.002	Gist 1987; SAIC 1993
Ingestion rate of feed (dry weight) by dairy cattle (kg d^{-1})	$Q_{feed(d)}$	Triangular	! ! !	! ! !	10	! ! !	7	14	Shor & Fields 1980; Dreicer et al. 1990
Fraction of feed consumed by dairy cattle that was pasture (unitless)	$f_{p(d)}$	Triangular	! ! !	! ! !	0.60	! ! !	0.4	0.75	Shor & Fields 1980; Koranda 1965
Biotransfer factor from cattle intake (pasture) to milk $(mg\ L^{-1})/(mg\ d^{-1})$	$F_{m(p)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	5×10^{-6}	5×10^{-5}	Mullen et al. 1975; Neathery et al. 1974; Potter et al. 1972
Ingestion rate of milk (L d^{-1}) - *Adult female*	$U_{milk(a)}$	Triangular	! ! !	0.28	! ! !	! ! !	0.12	0.85	USDA, 1955a, 1955b, 1966
Ingestion rate of milk (L d^{-1}) -*Child (6 mo-3 yrs)*	$U_{milk(c)}$	Triangular	! ! !	0.71	! ! !	! ! !	0.24	1.2	USDA, 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of milk consumed that was home-produced (unitless)	f_{mh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Soil ÷ Pasture ÷ Livestock ÷ Beef ÷ Humans [Adult (a) and Child(c)]	$Dose_{soil\&past\&beef} = (C_{soil} \times B_{past} \times Q_{feed(b)} \times f_{p(b)} \times F_{f(p)} \times U_{beef(a\ or\ c)} \times B_{oral\&food} \times f_{bh}) / BW_{(a\ or\ c)}$								
Concentration ratio for transfer of mercury from dry soil to pasture (dry weight) (unitless)	B_{past}	Log-triangular	! ! !	! ! !	0.0005	! ! !	0.0001	0.002	Gist 1987; SAIC 1993
Ingestion rate of feed (dry weight) by beef cattle (kg d^{-1})	$Q_{feed(b)}$	Triangular	! ! !	! ! !	9	! ! !	6	13	Baes et al. 1984; Mayland et al. 1977; NCRP 1985; Ng et al. 1978; Whicker & Kirchner 1987
Fraction of feed consumed by beef cattle that was pasture (unitless)	$f_{p(b)}$	Triangular	! ! !	! ! !	0.6	! ! !	0.4	0.75	Shor & Fields 1980
Biotransfer factor from cattle intake (pasture) to beef (mg kg^{-1})/(kg d^{-1})	$F_{f(p)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	0.0001	0.0009	Vreman et al. 1986
Ingestion rate of beef (kg d^{-1}) - *Adult female*	$U_{beef(a)}$	Triangular	! ! !	0.10	! ! !	! ! !	0.032	0.25	USDA 1955a, 1955b, 1966
Ingestion rate of beef (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{beef(c)}$	Triangular	! ! !	0.039	! ! !	! ! !	0.010	0.11	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of beef consumed that was home-produced (unitless)	f_{bh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966
Sediment ÷ Humans (Ingestion) [Adult (a) and Child (c)]	$Dose_{sed(ing)} = (C_{sed} \times U_{sed(a\ or\ c)} \times B_{oral\&soil} \times f_{sc(a\ or\ c)} \times EF_{EF(a\ or\ c)}) / BW_{(a\ or\ c)}$								
Ingestion rate of sediment (kg d^{-1}) - *Adult female*	$U_{sed(a)}$	Lognormal	! ! !	0.000025	! ! !	0.000020	! ! !	! ! !	Finley et al. 1994
Ingestion rate of sediment (kg d^{-1}) - *Child*	$U_{sed(c)}$	Lognormal	! ! !	0.000075	! ! !	0.000060	! ! !	! ! !	Finley et al. 1994
Relative bioavailability of mercury in soil following ingestion (unitless)	$B_{oral\text{-}soil}$	Lognormal	! ! !	0.053	! ! !	0.10	! ! !	! ! !	Barnett & Turner 1995

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Fraction of sediment ingested that was contaminated (unitless) - *Adult female*	$f_{sc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Fraction of sediment ingested that was contaminated (unitless) -*Child (6 mo-3 yrs)*	$f_{sc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Adult female*	$EF_{sed(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.011	0.066	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Child (6 mo-3 yrs)*	$EF_{sed(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.0055	0.088	Prof. judgement
Sediment ÷ Humans (Dermal Contact) [Adult (a) and Child (c)]	$Dose_{sed(derm)} = (C_{sed} \times SA_{sed(a\ or\ c)} \times SL_{sed} \times B_{derm\&soil} \times f_{sc(a\ or\ c)} \times EF_{EF(a\ or\ c)} \times 10^{\&6}\ mg\ k$								
Surface area of exposed skin, dermal contact with soil (cm^2 d^{-1}) - *Adult female*	$SA_{soil(a)}$	Lognormal	! ! !	3100	! ! !	300	! ! !	! ! !	USEPA 1995a
Surface area of exposed skin, dermal contact with soil–Surface area-to-body weight ratio (cm^2 kg^{-1} d^{-1}) -*Child (6 mo-3 yrs)*	$SA_{soil(c)}$	Lognormal	! ! !	220	! ! !	33	! ! !	! ! !	USEPA 1995a
Sediment loading on skin (mg cm^{-2})	SL_{sed}	Lognormal	! ! !	0.52	! ! !	0.99	! ! !	! ! !	Finley et al. 1994
Relative bioavailability of mercury in sediment upon dermal contact (unitless)	$B_{derm\text{-}sed}$	Log-triangular	! ! !	0.023	! ! !	! ! !	0.006	0.11	USEPA 1994
Fraction of sediment dermally contacted that was contaminated (unitless)	f_{sc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Adult female*	$EF_{sed(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.011	0.066	Prof. judgement
Exposure frequency to sediment (d d^{-1}) - *Child (6 mo-3 yrs)*	$EF_{sed(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.0055	0.088	Prof. judgement

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Surface water ÷ Humans (Incidental ingestion) [Adult (a) and Child (c)]	$Dose_{water(inc\&ing)} = (C_{water} \times U_{water\&inc(a\ or\ c)} \times B_{oral\&water} \times f_{wc} \times ET_{EF(a\ or\ c)} \times EF_{EF(a\ or\ c)}) / BW_{(a\ or\ c)}$								
Incidental ingestion rate of surface water (L h^{-1}) - *Adult female*	$U_{water\text{-}inc(a)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0	0.025	USEPA 1989
Incidental ingestion rate of surface water (L h^{-1}) -*Child (6 mo-3 yrs)*	$U_{water\text{-}inc(c)}$	Uniform	! ! !	! ! !	! ! !	! ! !	0	0.05	USEPA 1989
Relative bioavailability of mercury in water following ingestion (unitless)	$B_{oral\text{-}water}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Fraction of surface water ingested that was contaminated (unitless)	f_{wc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure time to surface water (h d^{-1}) - *Adult female*	$ET_{EF(a)}$	Triangular	! ! !	! ! !	0.25	! ! !	0.08	2.0	Prof. judgement
Exposure time to surface water (h d^{-1}) - *Child (6 mo-3 yrs)*	$ET_{EF(c)}$	Triangular	! ! !	! ! !	0.33	! ! !	0.08	3.0	Prof. judgement
Exposure frequency to surface water (d d^{-1}) - *Adult female*	$EF_{EF(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.011	0.066	Prof. judgement
Exposure frequency to surface water (d d^{-1}) -*Child (6 mo-3 yrs)*	$EF_{EF(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.0055	0.088	Prof. judgement
Surface water ÷ Humans (Dermal contact) [Adult (a) and Child (c)]	$Dose_{water(derm)} = (C_{water} \times SA_{w(a\ or\ c)} \times PC \times f_{wc} \times ET_{EF(a\ or\ c)} \times EF_{EF(a\ or\ c)} \times 10^{\&3}\ L\ cm^{\&3} \times BW_{(c)}\ [child\ only]) / BW_{(a\ or\ c)}$								
Surface area of exposed skin, dermal contact with surface water (cm^2) - *Adult female*	$SA_{w(a)}$	Lognormal	! ! !	8,000	! ! !	800	! ! !	! ! !	USEPA 1995a
Surface area of exposed skin, dermal contact with surface water– Surface area-to-body weight ratio (cm^2 kg^{-1}) - *Child (6 mo-3 yrs)*	$SA_{w(c)}$	Lognormal	! ! !	400	! ! !	100	! ! !	! ! !	USEPA 1995a
Permeability constant for mercury (cm h^{-1})	PC	Loguniform	! ! !	! ! !	! ! !	! ! !	0.001	0.01	USEPA 1994

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Fraction of surface water dermally contacted that was contaminated (unitless)	f_{wc}	Uniform	! ! !	! ! !	! ! !	! ! !	0.8	1.0	Prof. judgement
Exposure time to surface water (h d^{-1}) - *Adult female*	$ET_{sed(a)}$	Triangular	! ! !	! ! !	0.25	! ! !	0.08	2.0	Prof. judgement
Exposure time to surface water (h d^{-1}) - *Child (6 mo-3 yrs)*	$ET_{sed(c)}$	Triangular	! ! !	! ! !	0.33	! ! !	0.08	3.0	Prof. judgement
Exposure frequency to surface water (d d^{-1}) - *Adult female*	$EF_{sed(a)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.011	0.066	Prof. judgement
Exposure frequency to surface water (d d^{-1}) -*Child (6 mo-3 yrs)*	$EF_{sed(c)}$	Triangular	! ! !	! ! !	0.044	! ! !	0.0055	0.088	Prof. judgement
Surface water ÷ Dairy cows ÷ Milk ÷ Humans [Adult (a) and Child(c)]	$Dose_{water\&milk} = (C_{water} \times Q_{water(d)} \times f_{wdc} \times F_{m(w)} \times U_{milk(a\ or\ c)} \times B_{oral\&food} \times f_{mh}) / BW_{(a\ or\ c)}$								
Ingestion of surface water by dairy cattle (L d^{-1})	$Q_{water(d)}$	Uniform	! ! !	! ! !	! ! !	! ! !	32	60	McKone 1988
Fraction of water ingested by dairy cattle that was from East Fork Poplar Creek (unitless)	f_{cw}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Biotransfer factor from cattle intake (water) to milk (mg L^{-1})/(mg d^{-1})	$F_{m(p)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	5×10^{-6}	5×10^{-5}	Mullen et al. 1975; Neathery et al. 1974; Potter et al. 1972
Ingestion rate of milk (L d^{-1}) - *Adult female*	$U_{milk(a)}$	Triangular	! ! !	0.28	! ! !	! ! !	0.12	0.85	USDA 1955a, 1955b, 1966
Ingestion rate of milk (L d^{-1}) -*Child (6 mo-3 yrs)*	$U_{milk(c)}$	Triangular	! ! !	0.71	! ! !	! ! !	0.24	1.2	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of milk consumed that was home-produced (unitless)	f_{mh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement
Surface water ÷ Livestock ÷ Beef ÷ Humans [Adult (a) and Child(c)]	$Dose_{water\&beef} = (C_{water} \times Q_{water(b)} \times f_{wbc} \times F_{f(w)} \times U_{beef(a\ or\ c)} \times B_{oral\&food} \times f_{bh}) / BW_{(a\ or\ c)}$								

TABLE 9-4: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE EAST FORK POPLAR CREEK FARM FAMILY POPULATION

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Ingestion of surface water by beef cattle (L d^{-1})	$Q_{water(b)}$	Uniform	! ! !	! ! !	! ! !	! ! !	22	50	McKone 1988
Fraction of water ingested by beef cattle that was from EFPC (unitless)	f_{wbc}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Biotransfer factor from cattle intake (water) to beef (mg L^{-1})/(mg d^{-1})	$F_{f(w)}$	Loguniform	! ! !	! ! !	! ! !	! ! !	0.0001	0.0009	Vreman et al. 1986
Ingestion rate of beef (kg d^{-1}) - *Adult female*	$U_{beef(a)}$	Triangular	! ! !	0.10	! ! !	! ! !	0.032	0.25	USDA 1955a, 1955b, 1966
Ingestion rate of beef (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{beef(c)}$	Triangular	! ! !	0.039	! ! !	! ! !	0.010	0.11	USDA 1955a, 1955b, 1966
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement
Fraction of beef consumed that was home-produced (unitless)	f_{bh}	Uniform	! ! !	! ! !	! ! !	! ! !	0.7	1.0	USDA 1955a, 1955b, 1966; Prof. judgement
Fish ÷ Humans (Ingestion) [Adult (a) and Child (c)]	$Dose_{fish} = (C_{fish} \times U_{fish(a\ or\ c)} \times B_{oral\&food} \times f_{fc(a\ or\ c)}) / BW_{(a\ or\ c)}$								
Ingestion rate of fish from East Fork Poplar Creek (kg d^{-1}) - *Adult female*	$U_{fish(a)}$	Lognormal	! ! !	0.0012	! ! !	2.9	! ! !	! ! !	Ebert 1996
Ingestion rate of fish from East Fork Poplar Creek (kg d^{-1}) - *Child (6 mo-3 yrs)*	$U_{fish(c)}$	Custom	Adult fish consumption rate (kg d^{-1}) ×Child body weight (kg) × 1.3 / 70 kg						Ebert 1996
Relative bioavailability of methylmercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement

a Arithmetic mean
b Arithmetic standard deviation

TABLE 9-5: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE OAK RIDGE COMMUNITY– NEAR-FLOODPLAIN RESIDENTS

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
General Exposure Assumptions									
Body weight (kg)- *Adult female*	$BW_{(a)}$	Lognormal	! ! !	62	! ! !	9.5	! ! !	! ! !	USEPA 1995a
Body weight (kg)- *Child (6 mo-3 yrs)*	$BW_{(c)}$	Normal	! ! !	12	! ! !	2.2	! ! !	! ! !	USEPA 1995a
Air ÷ Humans (Inhalation) [Adult (a) and Child (c)] $Dose_{air(inh)} = (C_{air} \times U_{air(a\ or\ c)} \times f_h \times [f_{ho(a\ or\ c)} + ((1 - f_{ho(a\ or\ c)}) \times r_{io})] \times B_{inh}) / BW_{(a\ or\ c)}$									
Inhalation rate ($m^3\ d^{-1}$)- *Adult female*	$U_{air(a)}$	Lognormal	! ! !	17	! ! !	3.2	! ! !	! ! !	Layton 1993
Inhalation rate ($m^3\ d^{-1}$)-*Child (6 mo-3 yrs)*	$U_{air(c)}$	Lognormal	! ! !	5.9	! ! !	1.1	! ! !	! ! !	Layton 1993
Fraction of time at home (unitless)- *Adult female*	$f_{h(a)}$	Triangular	! ! !	! ! !	0.88	! ! !	0.68	0.98	Prof. judgement
Fraction of time at home (unitless)- *Child (6 mo- 3 yrs)*	$f_{h(c)}$	Triangular	! ! !	! ! !	0.96	! ! !	0.88	1.0	Prof. judgement
Fraction of time at home outdoors (unitless) - *Adult female*	$f_{ho(a)}$	Triangular	! ! !	! ! !	0.14	! ! !	0.071	0.25	Prof. judgement
Fraction of time at home outdoors (unitless) -*Child (6 mo-3 yrs)*	$f_{ho(c)}$	Triangular	! ! !	! ! !	0.087	! ! !	0.022	0.22	Prof. judgement
Indoor-to-outdoor ratio (unitless)	r_{io}	Uniform	! ! !	! ! !	! ! !	! ! !	0.3	0.95	
Relative bioavailability of mercury in air following inhalation (unitless)	B_{inh}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Air ÷ Above-ground Fruits and Vegetables ÷ Humans [Adult (a) and Child(c)] $Dose_{air\&veg} = (C_{air} \times V_{D(veg)} \times \left(\frac{1 - e^{-k_w T_{g(v)}}}{k_w}\right) \times f_w \times U_{veg\&e(a\ or\ c)} \times B_{oral\&food}) / BW_{(a\ or\ c)}$									
Total deposition onto above-ground fruits and vegetables ($m\ d^{-1}$)/($m^2\ kg^{-1}$)	$V_{D(veg)}$	Lognormal	! ! !	124	! ! !	99	! ! !	! ! !	Calculated: See Sect. 8.1

TABLE 9-5: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE OAK RIDGE COMMUNITY– NEAR-FLOODPLAIN RESIDENTS

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
Weathering rate for vegetables (d^{-1})	k_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.05	0.12	Miller & Hoffman 1979
Period of exposure of standing crop biomass for vegetables (d)	$T_{g(v)}$	Uniform	! ! !	! ! !	! ! !	! ! !	10	90	Prof. judgement
Fraction of contaminant remaining after washing (unitless)	f_w	Uniform	! ! !	! ! !	! ! !	! ! !	0.5	1.0	IAEA 1992, 1994; Ng et al. 1978
Ingestion rate of homegrown above-ground fruits and vegetables (kg d^{-1}) - *Adult female*	$U_{veg\text{-}e(a)}$	Lognormal	! ! !	0.11	! ! !	0.12	! ! !	! ! !	USEPA 1995a
Ingestion rate of homegrown above-ground fruits and vegetables (kg d^{-1}) - *Child (6 mo-3 yrs)*	$U_{veg\text{-}e(c)}$	Lognormal	! ! !	0.056	! ! !	0.089	! ! !	! ! !	USEPA 1995a
Relative bioavailability of mercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Uniform	! ! !	! ! !	! ! !	! ! !	0.6	1.0	Prof. judgement

a Arithmetic mean
b Arithmetic standard deviation

TABLE 9-6: EXPOSURE EQUATIONS AND PARAMETER DISTRIBUTIONS FOR THE ANGLER POPULATIONS

Parameter	*Symbol*	*Distribution*	*Point Estimate or Custom*	*Mean* [a]	*Most Likely*	*Standard Deviation* [b]	*Minimum*	*Maximum*	*Reference*
General Exposure Assumptions									
Body weight (kg)- *Adult female*	$BW_{(a)}$	Lognormal	! ! !	62	! ! !	9.5	! ! !	! ! !	USEPA, 1995
Body weight (kg)- *Child (6 mo-3 yrs)*	$BW_{(c)}$	Normal	! ! !	12	! ! !	2.2	! ! !	! ! !	USEPA, 1995
Fish ÷ Humans (Ingestion) [Adult (a) and Child (c)]	$Dose_{fish}$ ' $(C_{fish} \times U_{fish(a\ or\ c)} \times B_{oral\&food} \times f_{fc(a\ or\ c)})$ / $BW_{(a\ or\ c)}$								
Ingestion rate of fish from Poplar Creek/ Clinch River- Commercial Angler (kg d^{-1}) -*Adult female*	$U_{fish(a)-}$	Lognormal	! ! !	0.0022	! ! !	0.0052	! ! !	! ! !	Ebert, 1996
Ingestion rate of fish from Poplar Creek/ Clinch River- Commercial Angler (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{fish(c)}$	Custom	Adult fish consumption rate (kg d^{-1}) ×Child body weight (kg) × 1.3 / 70 kg						Ebert, 1996
Ingestion rate of fish from Poplar Creek/ Clinch River- Recreational Angler (kg d^{-1}) -*Adult female*	$U_{fish(a)}$	Lognormal	! ! !	0.018	! ! !	0.043	! ! !	! ! !	Ebert, 1996
Ingestion rate of fish from Poplar Creek/ Clinch River- Recreational Angler (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{fish(c)}$	Custom	Adult fish consumption rate (kg d^{-1}) ×Child body weight (kg) × 1.3 / 70 kg						Ebert, 1996
Ingestion rate of fish from Watts Bar Reservoir- Commercial Angler (kg d^{-1}) -*Adult female*	$U_{fish(a)}$	Lognormal	! ! !	0.024	! ! !	0.057	! ! !	! ! !	Ebert, 1996
Ingestion rate of fish from Watts Bar Reservoir- Commercial Angler (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{fish(c)}$	Custom	Adult fish consumption rate (kg d^{-1}) ×Child body weight (kg) × 1.3 / 70 kg						Ebert, 1996
Ingestion rate of fish from Watts Bar Reservoir- Recreational Angler (kg d^{-1}) -*Adult female*	$U_{fish(a)}$	Lognormal	! ! !	0.030	! ! !	0.071	! ! !	! ! !	Ebert, 1996
Ingestion rate of fish from Watts Bar Reservoir- Recreational Angler (kg d^{-1}) -*Child (6 mo-3 yrs)*	$U_{fish(c)}$	Custom	Adult fish consumption rate (kg d^{-1}) ×Child body weight (kg) × 1.3 / 70 kg						Ebert, 1996
Relative bioavailability of methylmercury following ingestion of food (unitless)	$B_{oral\text{-}food}$	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement
Fraction of fish that is contaminated (unitless)	f_{cf}	Point	1.0	! ! !	! ! !	! ! !	! ! !	! ! !	Prof. judgement

a Arithmetic mean
b Arithmetic standard deviation

In this assessment, the relative bioavailability factor reflects the ratio of the percent bioavailability of mercury contacted through an environmental exposure (such as ingestion of mercury in soil), to the percent bioavailability of mercury in the study upon which the toxicity benchmark value is based. The derivation of relative bioavailability factors for each route of exposure addressed in the Task 2 assessment is discussed below.

9.2.1 Oral Bioavailability

As discussed in Section 5.2, oral exposure to mercury (i.e, through the ingestion route) is evaluated in this assessment for the following pathways and species of mercury:

- Ingestion of soil (inorganic mercury);
- Ingestion of fruits and vegetables contaminated through uptake of mercury from soil and air (inorganic mercury);
- Ingestion of milk contaminated through dairy cattle uptake of soil, air, and pasture (inorganic mercury);
- Ingestion of beef contaminated through beef cattle uptake of soil, air, and pasture (inorganic mercury); and,
- Ingestion of fish (methylmercury).

The PDFs used to describe the relative bioavailability factor for these pathways are presented in Table 9-7.

9.2.2 Bioavailability Following Dermal Contact with Soil

Prior to dermal absorption of mercury from soil, mercury must leach from the soil and be present on the skin in a form that can penetrate skin (e.g., dissolved in sweat). Metals that bind tightly to soil or are highly water-insoluble are less likely to desorb from soil into moisture on the skin (Horowitz and Finley 1994). Due to the strong adherence of mercury to soil and its low solubility, desorption of mercury from EFPC floodplain soil on the skin surface is likely to be extremely low.

Once mercury desorbs from soil on the skin surface, many factors influence absorption of mercury through the skin, including the age of the skin, skin condition, hydration, circulation to the skin, and skin temperature. The chemical structure and polarity of the compound also influence absorption. In general, polar compounds (i.e., compounds with a asymmetric electron distribution) are poorly absorbed through skin. Among the most polar compounds are those that spontaneously dissociate to form ions in an aqueous environment, including inorganic salts such as mercuric chloride (USEPA 1992b).

A literature search by project team investigators found no quantitative data on dermal absorption of inorganic mercuric salts or elemental mercury. In their recent Mercury Report to Congress, USEPA did not locate significant information on dermal absorption of mercury either (USEPA 1997). It is assumed that, because of the polarity of inorganic mercury compounds, they are unlikely to penetrate skin readily. Studies of the dermal absorption from soil of cadmium, another metal that forms polar inorganic compounds, indicate an absorption rate through human skin ranging from 0.08 to 1%, with an average absorption of 0.1% (USEPA 1992b). The percent absorption decreased with increasing concentrations.

There are no data on which to base a toxicity criterion for toxic effects due to dermal absorption of mercury. In the absence of data from dermal absorption studies, USEPA recommends evaluating toxicity due to dermal exposure using toxicity criteria for oral exposure and adjusting for differences in absorbed doses between the two modes of exposure using a relative bioavailability factor (USEPA 1989a). A relative dermal bioavailability factor ($B_{dermal\text{-}soil}$) for exposure to mercury in soil was calculated by comparing the approximated dermal bioavailability of mercury in soil to the oral bioavailability of mercuric chloride in water:

$$B_{dermal\&soil} = \frac{Dermal\ bioavailability\ of\ mercury\ in\ soil}{Oral\ bioavailability\ of\ mercuric\ chloride\ in\ water} \qquad (9.1)$$

Based on analogy to cadmium, dermal bioavailability of mercury in soil was characterized by a PDF with a minimum value of 0.1% and a maximum value of 1%. The distribution was assumed to be loguniform. This distribution was assumed to be reasonable because of the likelihood of:

- Limited desorption of inorganic mercury from EFPC floodplain soil on the skin surface;
- Limited absorption of desorbed mercury through skin; and,
- Decreasing rate of dermal absorption as soil mercury concentration increases.

Data on the oral bioavailability of mercuric chloride in water following ingestion by humans are not available. In laboratory animals, oral bioavailability of mercuric chloride in water ranged from 1% for adult mice to 38% for week old mice (ATSDR 1994). A USEPA expert panel recommended evaluating uptake of mercuric chloride from water following ingestion using a gastrointestinal absorption rate of 7% (Eastern Research Group 1988). Based on these data, the oral bioavailability of mercuric chloride in water was characterized by a PDF with a minimum value of 1%, a maximum value of 38%, and a central value of 7%. The distribution was assumed to be triangular.

The distribution used to characterize $B_{dermal\text{-}soil}$ was calculated by combining the distributions for dermal bioavailability of mercury in soil and oral bioavailability of mercuric chloride in water. Based on these calculations, $B_{dermal\text{-}soil}$ was characterized by a triangular distribution with a central value of 2.3% and minimum and maximum values (corresponding to the 5^{th} and 95^{th} percentiles of the distribution created by combining the dermal bioavailability and oral bioavailability distributions) of 0.6% and 11%, respectively.

9.2.3 Absorption Following Dermal Contact with Water

Contaminants may also be absorbed through skin from dermal contact with contaminated water. The skin permeability constant (K_p) describes absorption of a compound across skin from water. A number of studies of the absorption of aqueous solutions of mercuric mercury compounds across skin are described by USEPA (1992b). These studies estimate mercury absorption based on the "disappearance" of mercury applied to guinea pig skin *in vivo* or mass balance measurements of mercury applied to human skin *in vitro*. K_p estimates for inorganic mercury compounds (mercuric chloride and potassium mercuric iodide) from five studies range from 0.00093 cm h^{-1} to 0.011 cm h^{-1} (USEPA 1992b). Based on these values, K_p was characterized by a loguniform distribution with minimum and maximum values of 0.001 cm h^{-1} and 0.01 cm h^{-1}, respectively.

9.2.4 Bioavailability Following Inhalation of Vapor

Exposure to mercury in air was evaluated assuming inhalation of elemental mercury vapor. The inhalation reference concentration (RfC) for mercury is based on inhalation of mercury vapor by workers. Therefore, a relative bioavailability factor of 1.0 was assumed for the "inhalation of airborne mercury" pathway.

PDFs established to characterize the relative bioavailability of different mercury compounds in humans following ingestion, dermal contact, and inhalation are summarized in Table 9-7.

Table 9-7: Probability Density Functions for Characterizing Bioavailability in Humans

Parameter		PDF	
Symbol	**Description**	**Distribution**	**Description**
$B_{oral\text{-}food}$	Relative bioavailability of mercury in food following ingestion	Uniform	Minimum = 0.6 Maximum = 1.0
$B_{oral\text{-}water}$	Relative bioavailability of mercury in water following ingestion	Uniform	Minimum = 0.8 Maximum = 1.0
$B_{oral\text{-}soil}$	Relative bioavailability of mercury in soil following ingestion	Lognormal	Mean = 0.053 Standard deviation = 0.10
$B_{oral\text{-}fish}$	Relative bioavailability of mercury in fish following ingestion	Point estimate	Most likely = 1.0
$B_{derm\text{-}soil}$	Relative bioavailability of mercury in soil upon dermal contact	Logtriangular	Minimum = 0.006 Mode = 0.023 Maximum = 0.11
PC	Permeability constant for mercury in water	Loguniform	Minimum = 0.001 cm h^{-1} Maximum = 0.01 cm h^{-1}
B_{inh}	Relative bioavailability of mercury in air following inhalation	Point estimate	Most likely = 1.0

10.0 ESTIMATION OF DOSES TO POTENTIALLY EXPOSED POPULATIONS

This section presents summaries of results of the dose reconstruction for each population of interest and for the key exposure pathways. Detailed tables of estimated mercury doses are presented in Appendix W. For each population of interest and exposure pathway evaluated, central estimate (50^{th} percentile) doses, along with 95% lower (2.5^{th} percentile) and upper (97.5^{th} percentile) confidence limits about the central estimate are presented for each year of interest. For most populations of interest evaluated in this assessment, exposure to inorganic mercury was assumed to be associated with more than one pathway; consequently, *total* inorganic mercury doses were also estimated by summing the estimated doses for the individual pathways.

For the Robertsville School children population, annual average daily doses of elemental mercury and inorganic mercury were estimated for children ages 12 to 15. For the Wolf Valley, Scarboro Community, EFPC Floodplain Farm Family, and Oak Ridge Community residents, annual average daily doses of elemental mercury and inorganic mercury were estimated for adult females of child-bearing age and for young children (assumed to be 6 months to 3 years of age). Although adult doses were estimated using intake rates for many parameters that are specific to adult females (such as vegetable consumption and inhalation rates), these estimated doses are assumed to also be applicable to adult males and adult females of different ages, because the estimated doses are normalized to body weight (they are presented in terms of milligrams of mercury taken in per kilogram of body weight per day, or mg kg^{-1} d^{-1}). In addition, for the fish consumer populations (who consumed fish from the Clinch River/ Poplar Creek or Watts Bar Reservoir systems) and members of the Scarboro Community and EFPC Floodplain Farm Family who caught and consumed fish from EFPC, annual average daily doses of methylmercury were estimated for adult females of child-bearing age (assumed to apply to both adult females and adult males) and young children (assumed to be 6 months to 3 years of age). As will be discussed in Sections 11 and 12, estimated methylmercury doses for adult females of child-bearing age were compared to toxicity benchmarks for both adult exposure and *in utero* exposure, since unborn children may be more sensitive to the effects of methylmercury exposure than adults.

The results presented in Appendix W include the following:

- Table W-1 (Page W-3) presents estimated annual average daily mercury doses (in units of mg kg^{-1} d^{-1}) to *Wolf Valley residents*, sometimes also called "downvalley" residents. Estimated doses are given for adults and children (age 6 months to 3 years), from inhalation of mercury (assumed to be elemental mercury) and three indirect pathways (assumed to be inorganic mercury) arising from the presence of airborne mercury. Results are given for 1953 through 1962, the years during which Colex processing was actively releasing airborne mercury from buildings on the Y-12 site. For each year, the estimated *total* inorganic mercury dose, representing the summation of doses from the three indirect pathways, is also presented.

- Table W-2 (Pages W-4 through W-14) presents estimated annual average daily mercury doses (in units of mg kg^{-1} d^{-1}) to residents of the *Scarboro Community* for 1950 through 1990. Results are given for adults and children (age 6 months to 3 years) for ten exposure pathways, including inhalation of mercury (assumed to be elemental mercury), consumption of fish from EFPC (assumed to be methylmercury), and eight pathways associated with exposure to inorganic mercury. For each year, the estimated *total* inorganic mercury dose, representing the summation of doses from the eight inorganic mercury pathways, is also presented.

- Table W-3 (Pages W-15 through W-18) presents estimated annual average daily mercury doses (in units of mg kg^{-1} d^{-1}) to *Robertsville School children* for 1950 through 1990. Results are given for seven exposure pathways, including inhalation of mercury (assumed to be elemental mercury) and six exposure pathways associated with exposure to inorganic mercury. For each year, the estimated *total* inorganic mercury dose, representing the summation of doses from the six inorganic mercury pathways, is also presented.

- Table W-4 (Pages W-19 through W-29) presents estimated annual average daily mercury doses (in units of mg kg^{-1} d^{-1}) to adults and children of the *EFPC Floodplain Farm Family* for 1950 through 1990. Results are given for adults and children (age 6 months to 3 years) for 18 exposure pathways, including inhalation of mercury (assumed to be elemental mercury), consumption of fish from EFPC (assumed to be methylmercury), and 16 pathways associated with exposure to inorganic mercury. For each year, the estimated *total* inorganic mercury dose, representing the summation of doses from the 16 inorganic mercury pathways, is also presented.

- Tables W-5 and W-6 (Pages W-30 through W-32 and W-33 through W-35, respectively) present estimated annual average daily mercury doses (in units of mg kg^{-1} d^{-1}) to *Oak Ridge Community Populations 1 and 2*, respectively, for 1950 through 1990. Results are given for adults and children (age 6 months to 3 years), for inhalation of mercury (assumed to be elemental mercury) and consumption of vegetables contaminated from airborne mercury (assumed to be present in the plants as inorganic mercury), in both cases due to volatilization of mercury from EFPC.

- Table W-7 (Pages W-36 through W-41) presents estimated annual average daily methylmercury doses (in units of mg kg^{-1} d^{-1}) to three categories of fish consumers who ate fish from the Clinch River/ Poplar

Creek and Watts Bar Reservoir systems. The three categories were subjectively chosen based on the number of fish meals consumed per year. In addition, this table presents estimated doses for Category 3 anglers in EFPC (it is not likely that Category 1 and 2 anglers, who consumed a larger number of fish meals, existed for EFPC) and four populations of anglers downstream of Oak Ridge, namely recreational and commercial anglers in the Clinch River/Poplar Creek system and in Watts Bar Reservoir for 1950 through 1990.

Peak estimated mercury doses of each mercury species (summed across all relevant exposure pathways) to each population of interest evaluated in the Task 2 mercury dose reconstruction are shown in Figures 10-1 through 10-3. For each population of interest, the central estimate (50th percentile) of the peak dose is shown, along with the 95% lower and upper confidence bounds.

Figures 10-1 through 10-3 show:

- For all populations of interest, the highest annual average doses were estimated to have occurred during the mid- to late-1950s. These were the years of highest releases of mercury from Y-12 to air and to EFPC.

- Estimated annual average methylmercury doses to individuals who consumed fish from Watts Bar Reservoir were about 4-fold lower than doses estimated for individuals who consumed the same amount of fish from Clinch River/ Poplar Creek during the same year.

- With the exception of exposures of fish consumers to methylmercury in fish, estimated doses to the EFPC Floodplain Farm Family population are the highest of all exposure populations that were evaluated.

- Estimated peak total inorganic mercury doses to Wolf Valley (“downvalley”) residents, resulting from direct air releases of mercury from Y-12, are about 30- to 40-fold lower than peak doses estimated for the EFPC Floodplain Farm Family.

- Estimated peak inhalation doses to Scarboro residents are about 9-fold lower than doses estimated for the EFPC Floodplain Farm Family, due in part to the greater distance of the Scarboro Community from EFPC.

Figure 10-1: Peak Estimated Total Doses of Inorganic Mercury;
Annual Average Daily Doses Calculated for Each Population of Interest

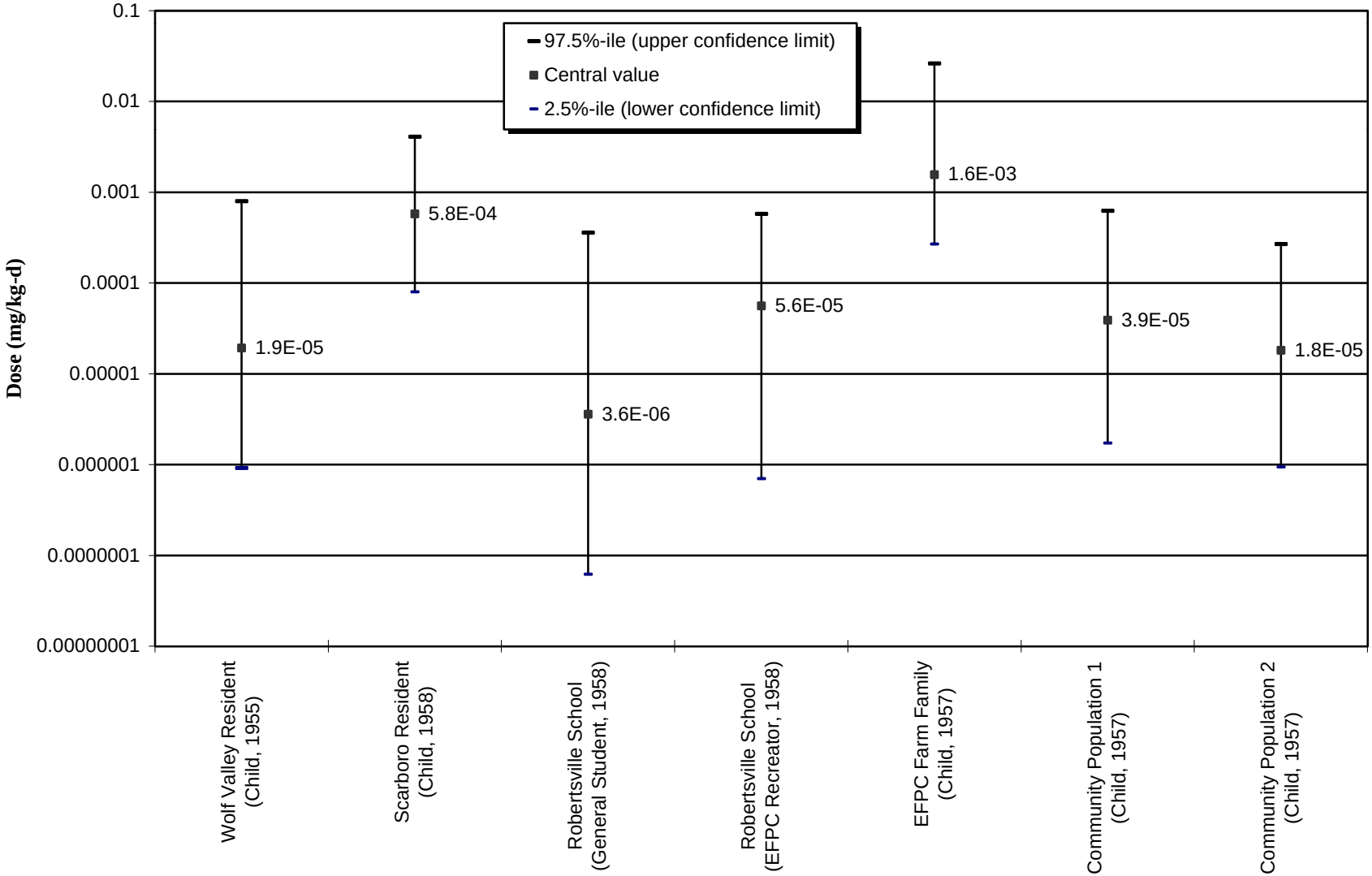


Figure 10-2: Peak Estimated Total Doses of Elemental Mercury;
Annual Average Daily Doses Calculated for Each Population of Interest

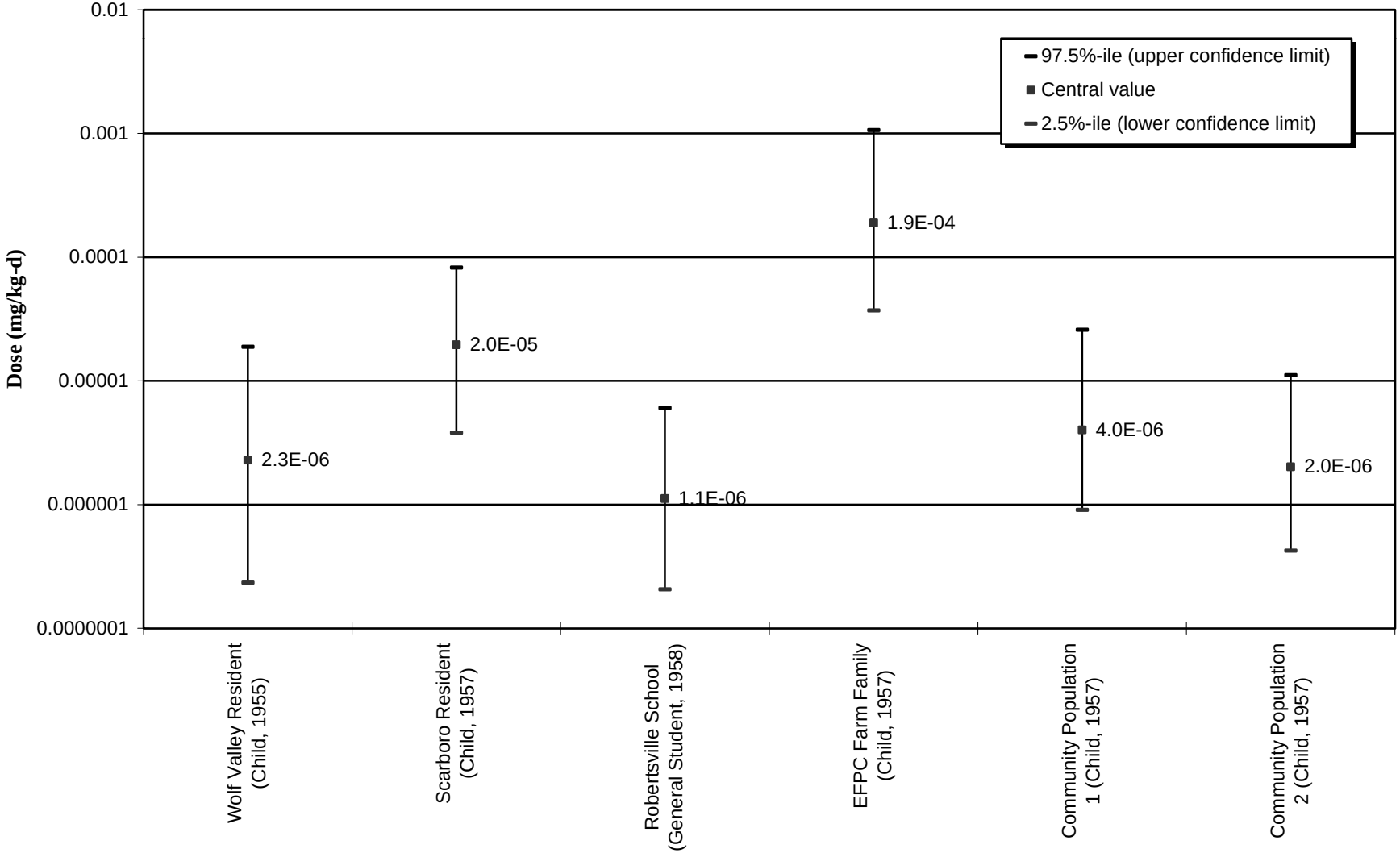


Figure 10-3: Peak Estimated Total Doses of Methylmercury;
Annual Average Daily Doses Calculated for Each Population of Interest

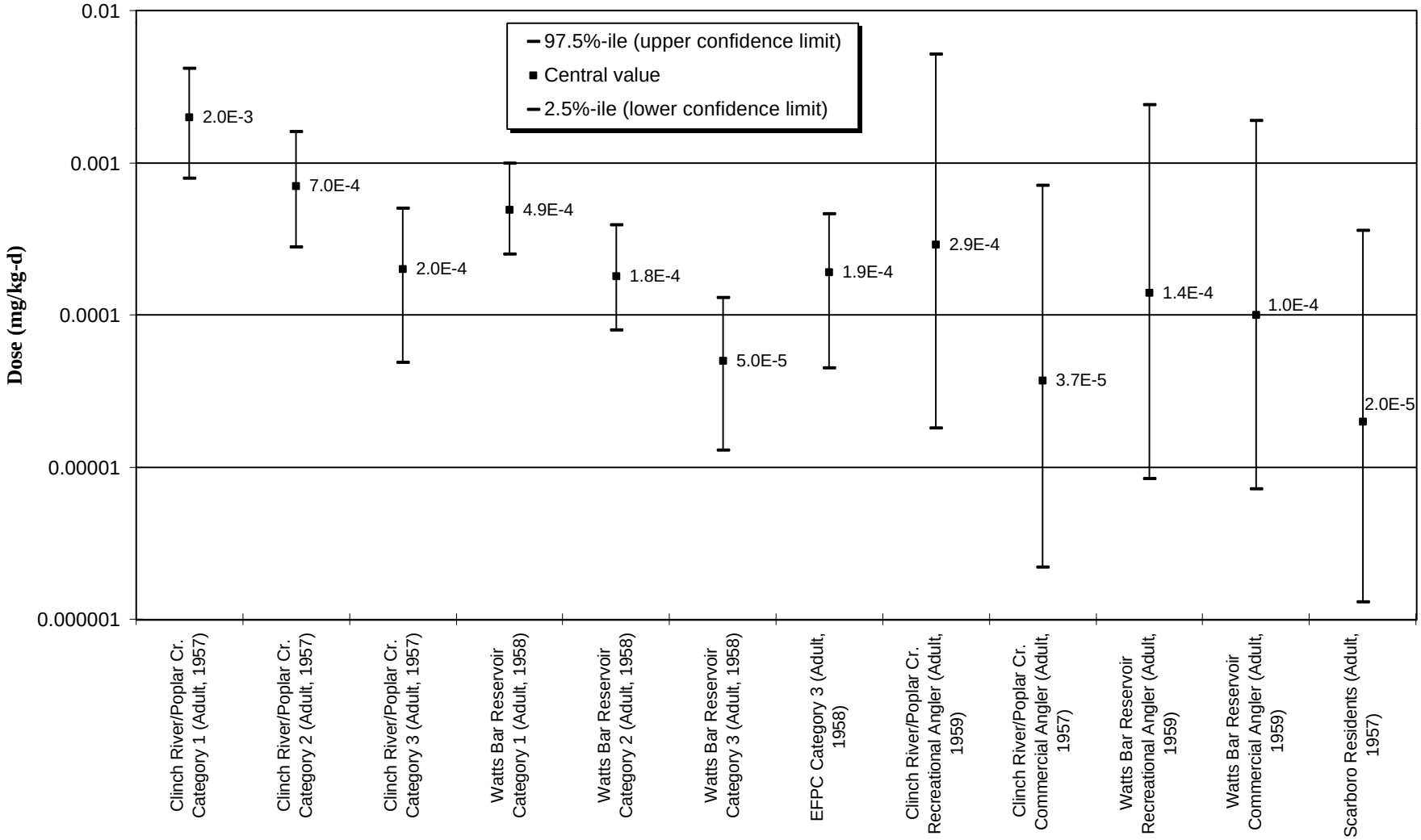


Figure 10-4 illustrates the predominant exposure pathways contributing to the estimated total inorganic mercury dose for each population of interest depicted in Figure 10-1. Contributions to total dose presented in these charts are based on central estimate values. The following conclusions can be made concerning the relative contribution of individual exposure pathways to the total inorganic mercury dose for the six populations of interest:

- Estimated total inorganic mercury doses to Wolf Valley ("downvalley") residents are dominated by consumption of vegetables contaminated from airborne mercury from the Y-12 Plant.

- Estimated total inorganic mercury doses to Scarboro residents are also dominated by consumption of vegetables contaminated from airborne mercury. Skin contact with waterborne mercury and incidental ingestion of waterborne mercury are the second most important pathways for Scarboro residents.

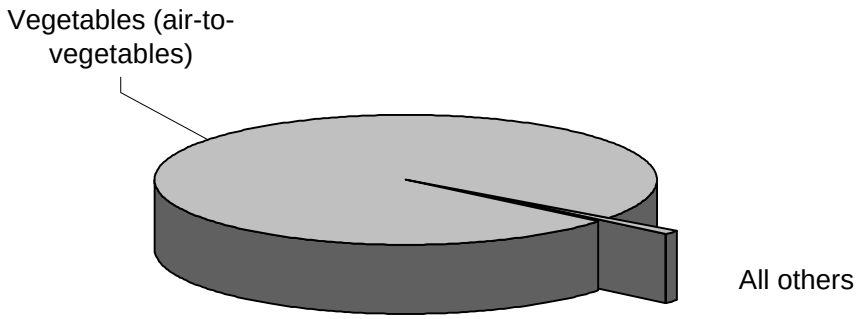
- Estimated total inorganic mercury doses to Robertsville General Students are dominated by skin contact with contaminated soil. For the Robertsville School Student Recreator, inorganic mercury doses are equally contributed by skin contact with contaminated soil and skin contact with waterborne mercury.

- Estimated total inorganic mercury doses to the EFPC Floodplain Farm Family population are dominated by consumption of vegetables contaminated from airborne mercury.

- Estimated total inorganic mercury doses to Community Populations 1 and 2, for which exposures from airborne mercury volatilized from EFPC were evaluated, are solely due to consumption of vegetables contaminated from airborne mercury.

Figure 10-4 (Page 1 of 2): Contribution of Individual Pathways to the Estimated Total Inorganic Mercury Dose (based on the average estimated dose for the highest year)

Wolf Valley Resident (Child, 1955)



Scarboro Resident (Child, 1955)

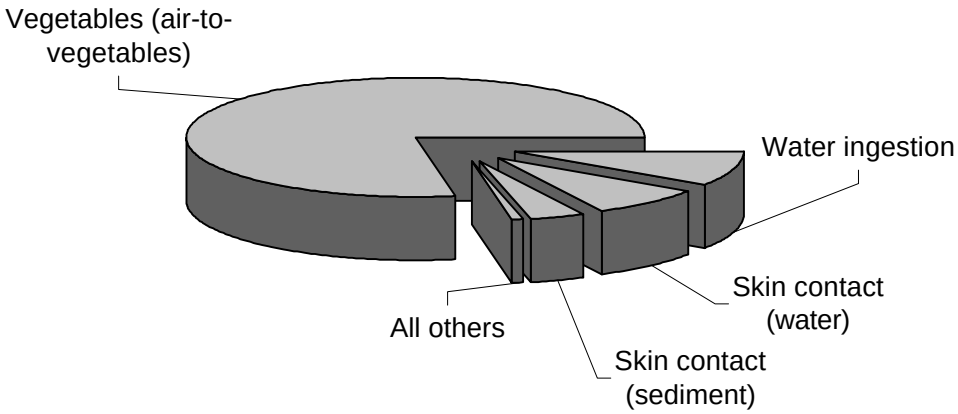
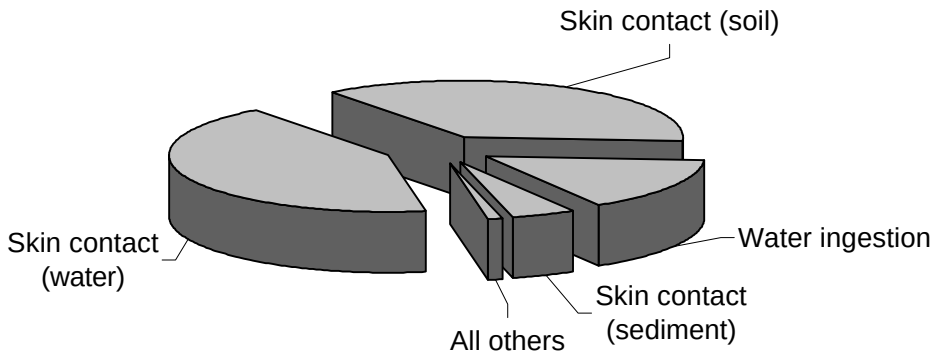
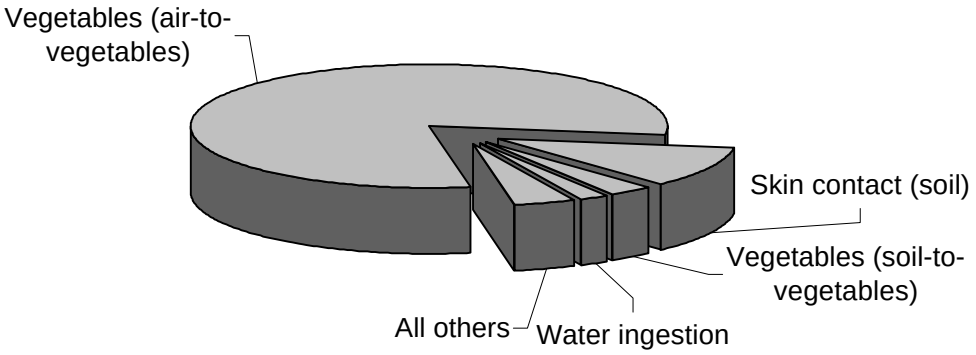


Figure 10-4 (Page 2 of 2): Contribution of Individual Pathways to the Estimated Total Inorganic Mercury Dose (based on the average estimated dose for the highest year)

Robertsville School- Student Recreator (1958)



EFPC Floodplain Farm Family (Child, 1958)



This page intentionally left blank.

11.0 TOXICITY BENCHMARKS FOR COMPARISON TO ESTIMATED MERCURY DOSES

The project team carefully reviewed studies investigating the toxicity of different species of mercury through various routes of exposure. Based on these studies, the project team identified toxicity benchmarks for comparison to the mercury doses that were estimated for members of the public who historically lived downwind or downstream of Y-12.

This section discusses the approach used to identify toxicity benchmarks and describes toxicological studies of the following species and routes of mercury exposure:

- Ingestion of inorganic mercury
- Inhalation of elemental mercury
- Ingestion of methylmercury

Then, toxicity benchmarks for each of these species of mercury and routes of exposure are determined. Threshold doses may vary among different individuals within a population; therefore, in this report, two different types of toxicity benchmarks were determined, to provide lower- and upper-bound estimates of doses that may potentially be associated with an increased incidence of adverse health effects. Lower-bound toxicity benchmarks were based on USEPA and ATSDR recommended levels of concern, which incorporate uncertainty or safety factors to account for uncertainties in application of these levels to exposed populations. Upper-bound toxicity benchmarks were based on no observed adverse effect levels, or NOAELs, which are the maximum doses at which no effects have been observed in laboratory animals or humans.

11.1 Determination of Toxicity Benchmarks

Mercury is typically evaluated as a systemic toxicant (that is, for noncarcinogenic endpoints) since there are no definitive data suggesting that any form of mercury is carcinogenic through any route of exposure. The approach used by the USEPA and other regulatory agencies to assess risks associated with systemic toxicity is to identify an exposure threshold below which adverse effects are not observed. The first adverse effect that occurs as the dose or concentration increases beyond the threshold is called the "critical effect" (Dourson et al. 1996). Selection of regulatory levels is generally based on the assumption that if the critical effect is prevented, then all toxic effects are prevented. Not all effects are adverse effects, and the judgement of what constitutes an adverse effect is sometimes difficult.

Typically, threshold doses for noncarcinogens are based on NOAELs (no observed adverse effect levels), or LOAELs (lowest observed adverse effect levels) if a NOAEL is not available. A NOAEL is the *highest* dose in a given study at which *no* statistically or biologically significant indication of the toxic effect of concern has been identified, while a LOAEL is the *lowest* dose at which the toxic effect has been identified. NOAELs and LOAELs are typically established either from investigations of adverse effects following past worker exposures, or studies of the prevalence of adverse effects in laboratory animals

exposed to defined dose levels. The advantage of animal studies is that dose levels are usually known and can be closely monitored, allowing more precise characterization of NOAELs and LOAELs. They are also often the only type of toxicity studies available for chemicals that are not typically used in the workplace or not associated with repeated workplace exposures.

Animal studies, however, introduce uncertainties in the extrapolation of doses from animals to humans. Worker studies, when available, are often preferable since they eliminate these type of uncertainties. In many worker studies, however, chronic exposures have continued for many years and exact dose levels in the past (such as air concentrations) and their effect on current symptoms are not known. In addition, it can be difficult to account for the effects of confounding variables (including such factors as alcohol consumption, lifestyle habits, caffeine consumption, and health status) on the results of the test being used to evaluate evidence of adverse health effects. Further, workers are typically a highly selected group with respect to age, sex, and general health, who may be less sensitive to adverse effects of exposure than other members of the general public (including children or the elderly).

Identification of a threshold dose or NOAEL for noncarcinogens assumes that there is a range of exposures from zero to some finite value that can be tolerated by the organism (either a human or a laboratory animal) with essentially no chance of expression of an adverse effect (Barnes and Dourson 1988). Threshold doses may vary among different individuals within a population. For this reason toxicity benchmark values established by regulatory agencies, such as those established by the USEPA or the ATSDR, are generally selected to keep exposures below the population threshold, defined as the lowest of the thresholds for individuals within a population (Barnes and Dourson 1988). To reflect the uncertainty in the data used to derive the threshold value, these recommended toxicity benchmarks incorporate safety or uncertainty factors. For example, a 3- to 10-fold uncertainty factor is typically used to address the extrapolation from animal to human doses, and a 3- to 10-fold uncertainty factor is used to extrapolate from an average human NOAEL to a sensitive human NOAEL. In addition, a 3- to 10-fold uncertainty factor is often used to address the extrapolation from a LOAEL to a NOAEL when a NOAEL is not available, since a LOAEL does not give a clear threshold below which adverse effects do not occur (because it is often not known how much lower than the LOAEL the dose must be decreased before the effect will no longer occur).

Both the USEPA and the ATSDR have established toxicity benchmarks for exposure to different mercury compounds. For example, the USEPA has established reference doses (RfDs) or reference concentrations (RfCs) for inorganic, elemental, and methylmercury. These RfDs or RfCs are identified as estimates (with uncertainty spanning perhaps an order of magnitude) of daily exposure to the human population (including sensitive subgroups such as children or the elderly) that are likely to be without an appreciable risk of deleterious effects during a lifetime (Barnes and Dourson 1988). RfDs (for oral exposure) are expressed in units of milligrams per kilogram of body weight per day (mg kg^{-1} d^{-1}). RfCs (for inhalation exposure) are expressed in units of milligrams per cubic meter of inhaled air (mg m^{-3}). Minimal Risk Levels (MRLs) developed by the ATSDR are not established to support regulatory action, but to acquaint health professionals with exposure levels at which adverse health effects are not expected to occur in humans. MRLs are expressed in units of mg kg^{-1} d^{-1}.

The following sections describe the data that are available for establishing dose-response relationships and toxicity benchmarks for exposure to different forms of mercury at low dose environmental exposures.

11.2 Summary of Dose-Response Data for Inorganic Mercury

As discussed in Section 5, mercury in soil, water, and vegetation near the ORR was likely present as a mixture of soluble and insoluble inorganic mercury compounds. For purposes of evaluating exposures to mercury in soil, water, and vegetation near the ORR, estimated doses were compared to toxicity benchmarks based on exposure to ingested inorganic mercury.

Studies of toxicological effects following exposure to inorganic mercury, and selection of toxicity benchmarks for comparison to estimated doses, are described below.

Toxicokinetics of Ingested Inorganic Mercury

When ingested, soluble inorganic mercury compounds (such as mercuric chloride, mercuric nitrate, and mercuric acetate) quickly dissociate into the mercuric cation (Hg^{2+}) and an associated anion, regardless of the original species (Gerstner and Huff 1977). Because this dissociation occurs so quickly, the patterns of tissue distribution, toxicity, and clearance (excretion from the body) are considered to be virtually identical for all soluble inorganic mercury compounds. Signs and symptoms of exposure to inorganic mercury compounds are primarily due to the Hg^{2+} cation; the associated anion (e.g., chloride, nitrate, acetate) is assumed to have only a minor influence on toxicity. After dissociating, Hg^{2+} ions enter reversible complexes with chemical groups on proteins and other compounds in the plasma or red blood cells (Gerstner and Huff 1977). Once bound, inorganic mercury does not readily penetrate "blood-brain" or placental barriers, such that functions of the central nervous system (CNS) and embryonic development remain relatively unaffected.

Exposure to high doses of inorganic mercury compounds, such as following administration of high doses to laboratory animals or one-time accidental or intentional ingestion of very large doses by humans, has been shown to cause injury to the GI tract and/or to be directly toxic to tubular lining cells in the kidney (Goyer 1996, Young 1991, Gerstner and Huff 1977). The kidneys generally exhibit the highest levels of mercury following exposure to large doses of inorganic mercury compounds (Young 1991). Toxic effects following low dose environmental exposures to inorganic mercury compounds have not been documented.

Dose-Response Relationships for Ingestion of Inorganic Mercury

Review of available literature shows that limited studies have been conducted evaluating the toxicity of inorganic mercury compounds. Although some data are available describing toxic effects following acute (short-term) exposures of humans to large quantities of inorganic mercury compounds (through accidental or purposeful ingestion of large quantities of mercury compounds), toxicological data on longer-term (subchronic or chronic) exposures to lower doses of inorganic mercury are generally limited to laboratory studies in which rodents are administered soluble inorganic mercury in a food or water matrix. As

discussed above, while these studies allow dose levels to be closely monitored so that NOAELs and LOAELs can be more precisely characterized, they introduce uncertainties associated with inter-species extrapolation.

Actual public exposures to inorganic mercury released from the ORR were not restricted to exposure to soluble forms through food or water ingestion. Rather, exposures likely involved ingestion of less soluble forms in soil as well as in food and water. It is not known with certainty what the actual speciation of mercury was historically in different environmental media near the ORR. For example, it is thought that much of the mercury in soil was in forms that are relatively insoluble. As discussed in Sections 5.2 and 9.2.1, acceptable exposure levels determined in animal studies using soluble forms of mercury would likely be lower than levels determined using insoluble forms, because the soluble forms are likely to be more bioavailable. However, toxicity studies using insoluble forms of inorganic mercury have not been conducted.

The USEPA RfD for inorganic mercury of 3×10^{-4} mg kg^{-1} d^{-1} is based on observations of kidney effects in rats administered water soluble mercuric chloride. This criterion was established by an expert panel from the USEPA drinking water equivalent level (DWEL) for inorganic mercury, using a weight-of-evidence approach. The panel, convened by the USEPA in 1987 to discuss key health and risk assessment issues pertaining to the oral intake of mercury, reviewed approximately 20 published studies addressing the toxicity of inorganic mercury. They evaluated studies in which the mercury was administered via several routes, including subcutaneous, intravenous, and ingestion (USEPA 1988), and selected three to support derivation of the DWEL (Druet et al. 1978, Bernaudin et al. 1981, Andres 1984).

All three studies evaluated kidney effects following administration of mercuric chloride to Brown Norway rats, which were considered to be a "good surrogate for the study of mercury-induced kidney damage in sensitive humans" (USEPA 1988). However, the route of administration for each study was different: Druet et al. (1978) administered mercuric chloride by subcutaneous injection, Bernaudin et al. (1981) by forcible feeding, and Andres (1984) by gavage. To extrapolate between routes of exposure, the USEPA panel recommended using a GI absorption rate of 7% and a subcutaneous absorption rate of 100% (USEPA 1988). The rationale for selection of a GI absorption rate of 7% was not presented and is not readily apparent. This is important because absorption was not measured in any of the key studies on which the inorganic mercury RfD was based and the bioavailability of soluble inorganic mercury salts in laboratory animals has been reported to be as high as 38%, although 7% may be a reasonable assumption of average bioavailability. In addition, in the study in which mercuric chloride was administered by forcible feeding (Bernaudin et al. 1981), no information on the vehicle in which mercuric chloride was administered was provided (for example, whether it was administered in food or water, or in some other medium).

The three studies support LOAELs for mercuric chloride administered via the oral route ranging from 0.23 to 0.63 mg kg^{-1} d^{-1}. The DWEL of 0.010 mg L^{-1} was established by multiplying an average LOAEL of 0.30 mg kg^{-1} d^{-1} by an average adult body weight of 70 kg, dividing by a drinking water ingestion rate of 2 L d^{-1}, and incorporating an uncertainty factor of 1000 (10 for extrapolation from a LOAEL to a NOAEL, 10 for use of subchronic studies, and a combined 10 for extrapolation from animals to humans and for

sensitive human populations). The RfD was in turn derived by multiplying the DWEL by a drinking water ingestion rate of 2 L d^{-1} and dividing by an average adult body weight of 70 kg.

ATSDR has developed Minimal Risk Levels (MRLs) for acute and intermediate level exposure to inorganic mercury of 7×10^{-3} mg kg^{-1} d^{-1} and 2×10^{-3} mg kg^{-1} d^{-1}, respectively. Both the acute and intermediate MRLs are based primarily on a study conducted by the National Toxicology Program (NTP) in which mercuric chloride was administered to rats and mice in water via gavage (through a tube into the stomach) at several dose levels for 16 days, 6 months, or 2 years (ATSDR 1994; NTP 1993).

In each study, mercury was administered 5 days per week for the duration of the study. The acute MRL is based on the NOAEL from the 16 day study of 0.93 mg kg^{-1} d^{-1}, while the intermediate MRL is based on the NOAEL from the 6 month study of 0.23 mg kg^{-1} d^{-1}. NOAELs from both the acute and intermediate exposure studies were based on renal effects (increased absolute and relative kidney weights) in rats. Acute and intermediate MRLs were derived by adjusting the NOAELs for continuous exposure (i.e., 7 days/week) and dividing by an uncertainty factor of 100 (10 for extrapolation from animals to humans and 10 for human variability). A chronic MRL was not developed from the results of the two-year chronic study because the study identifying the lowest LOAEL for chronic exposure observed a decreased survival rate for male rats (ATSDR 1994; ATSDR 1997).

Other Thresholds for Toxicological Effects

The LOAELs and NOAELs used to derive the USEPA and ATSDR toxicity criteria for inorganic mercury are all based on observations of adverse effects on the kidney, considered to be the critical effect for inorganic mercury exposure. There are no data on developmental or reproductive toxicity of inorganic mercury in humans following oral exposure, and only very limited data in animals. Studies of reproductive or developmental effects of inorganic mercury in animals suggest NOAELs for these effects are higher than NOAELs for kidney effects. For example, a study of developmental toxicity in hamsters following administration of mercuric chloride identified a NOAEL of 15.7 mg Hg/kg body weight (Young 1991, ATSDR 1994).

Selection of Toxicity Benchmarks for Comparison to Estimated Doses

The relevant data from the studies discussed above are summarized in Table 11-1.

Table 11-1: Animal Studies Used as the Basis for Establishing the USEPA RfD and the ATSDR MRLs for Inorganic Mercuric (Hg^{2+}) Chloride

Study	Animal Parameters [a]	Exposure Route	Duration (Days)	Toxicity Benchmarks (mg kg^{-1} d^{-1})
Druet et al. (1978)	Brown Norway rat (NA, M/F, 7-9 wks) [subchronic]	Subcutaneous	56	0.23 (LOAEL)[b]
Bernaudin et al. (1981)	Brown Norway rat (5, M/F, 8-10 wks) [subchronic]	Oral (forcible feeding)	60	0.32 (LOAEL)
Andres (1984)	Brown Norway rat (NA) [subchronic]	Gavage (in water)	60	0.63 (LOAEL)
NTP (1993)	F344 rat (10,M/F,6-7 wks) [acute]	Gavage (in water)	16	0.93[c] (NOAEL)
NTP (1993)	F344 rat (20, M/F, 6-7 wks) [intermediate]	Gavage (in water)	182	0.23 (NOAEL)

a The number of animals per dose level, sex, and age of test animals at beginning of study are summarized in parentheses; NA = Data not available

b Back-calculated from the subcutaneous route LOAEL of 0.016 mg kg^{-1} d^{-1} to an equivalent oral administered dose assuming the relative Hg^{2+} absorption is 7% for the oral route and 100% for the subcutaneous route.

c Acute exposure

As discussed above, the USEPA RfD and the ATSDR MRLs for ingestion of inorganic mercury range from 0.0003 to 0.002 mg kg^{-1} d^{-1}**S** these values incorporate safety factors to account for extrapolation of animal data to humans. LOAELs from studies in animals of duration from acute to subchronic range from 0.23 to 0.63 mg kg^{-1} d^{-1}, while NOAELs from acute and subchronic animal studies range from 0.23 to 0.93 mg kg^{-1} d^{-1}. While data are minimal, these data suggest that a NOAEL of approximately 0.1 mg kg^{-1} d^{-1} (calculated by dividing an average LOAEL by a LOAEL-to-NOAEL adjustment factor of 3) to 0.23 mg kg^{-1} d^{-1} is appropriate as an upper-bound toxicity benchmark for evaluating long-term exposures to inorganic mercury. These NOAELs, based on kidney effects, are expected to be lower than health effects thresholds for other toxicological endpoints, including developmental and reproductive effects.

11.3 Summary of Dose-Response Data for Elemental Mercury

As discussed in Sections 3, 4, and 5, mercury was released to air from ORR operations largely as elemental mercury vapor (UCCND 1983a), and mercury likely volatilized from both surface water and soil

(Lindberg et al. 1991; Xiao et al. 1991; Lindberg et al. 1995; Lindberg et al. 1996). Essentially all airborne mercury occurs as elemental mercury vapor (Lindberg et al., 1991; Fitzgerald and Clarkson 1991; Mason et al. 1994). Consequently, for purposes of evaluating exposures to airborne mercury near the ORR, estimated doses were compared to toxicity benchmarks based on elemental mercury vapor.

Studies of toxicological effects following exposure to elemental mercury, and selection of threshold levels for comparison to estimated doses, are described below.

Toxicokinetics of Inhaled Elemental Mercury

Inhaled elemental mercury vapor enters the bloodstream through the thin membranes of the lungs. In its elemental form, mercury readily penetrates the blood-brain barrier and accumulates in the CNS (Gerstner and Huff 1977). Acute exposure to very high concentrations of airborne elemental mercury has been associated with injury to the lungs, with symptoms resembling pneumonia, while the critical effect associated with chronic exposures to moderate air concentrations, such as may occur in the workplace, is considered to be CNS effects. Symptoms of elemental mercury toxicity following continued exposure to moderately high concentrations in air range from almost imperceptible disturbance of the CNS, including fine muscle tremors, insomnia, loss of appetite, and effects on the emotional state and memory, to incapacitation (Gerstner and Huff 1977; ATSDR 1994).

Dose-Response Data for Evaluating Airborne Elemental Mercury Exposures

In general, most of the data investigating adverse health effects associated with exposure to airborne elemental mercury are of two types:

- Studies investigating evidence of NOAELs or LOAELs in laboratory animals exposed to different doses of airborne elemental mercury vapor, and

- Worker exposure studies that investigate evidence of adverse health effects in workers occupationally exposed to elemental mercury vapor for many years.

Numerous investigators have examined health effects in workers occupationally exposed to elemental mercury vapor for many years. In a typical study, at least two exposure groups are examined: an exposed group and a control. Most studies include a questionnaire, in which the test subject provides information on employment history and lifestyle habits, as well as a clinical examination. Typical endpoints evaluated in the clinical examination include evidence of neurological effects (as measured by quantitative tests of tremor and neuromuscular functions or behavioral changes) or kidney dysfunction (as measured by analysis of proteins or other biochemical parameters in blood or urine). In addition, in the absence of, or to supplement air concentration data, mercury concentrations in blood or urine are often measured. However, concentrations in blood and urine reflect only recent exposures, while clinical manifestations may reflect exposures that occurred many years previously. Further, the relationship between blood or urine levels and exposure levels may vary considerably between individuals.

Dose-Response Relationships for Inhalation of Elemental Mercury

The USEPA RfC for inhalation of elemental mercury vapor of 3×10^{-4} mg m^{-3} is based on evidence of hand tremor, increases in memory disturbances, and slight subjective and objective evidence of autonomic dysfunction in workers occupationally exposed to elemental mercury (IRIS 1998). The RfC is based on six key studies (Fawer et al. 1983; Piikivi and Tolonen 1989; Piikivi and Hanninen, 1989; Piikivi 1989; Ngim et al. 1992; Liang et al. 1993). Three of these studies related neurological effects to time-weighted average (TWA) air concentrations measured in the workplace (Fawer et al. 1983; Ngim et al. 1992; Liang et al. 1993). Air concentrations were not measured in the remaining three studies (Piikivi and Tolonen 1989; Piikivi and Hanninen 1989; and Piikivi 1989). However, dose-response relationships were established by extrapolating from blood mercury concentrations to air concentrations, based on a conversion factor derived by Roels et al. (1987). The six studies used to derive the RfC suggest LOAEL air concentrations ranging from 0.023 to 0.033 mg m^{-3}.

The USEPA RfC was derived from a LOAEL of 0.025 mg m^{-3} by adjusting the 8-hour occupational exposure to continuous exposure assuming an occupational inhalation rate of 10 m^3 d^{-1} vs. a daily inhalation rate of 20 m^3 d^{-1} and a 5 d wk^{-1} occupational exposure vs. a continuous 7 d wk^{-1} exposure, then dividing by an uncertainty factor of 30 (a combined 10 for protection of sensitive human subpopulations and use of a LOAEL rather than a NOAEL, and 3 for limited data on developmental and reproductive studies):

$$0.025\ mg\ m^{-3} \times \frac{10\ m^3\ d^{-1}}{20\ m^3\ d^{-1}} \times \frac{5\ d\ wk^{-1}}{7\ d\ wk^{-1}} \approx 0.009\ mg\ m^{-3} \times \frac{1}{30} \approx 3 \times 10^{-4}\ mg\ m^{-3} \qquad (11.1)$$

An MRL of 0.0002 mg m^{-3} has been derived by ATSDR for chronic duration inhalation exposure to elemental mercury vapor (ATSDR 1997). The MRL is based on a significant increase, compared to controls, in the average velocity of naturally occurring tremors was observed in a group of 26 mercury-exposed workers from three industries exposed to low levels (0.026 mg m^{-3}) of mercury for an average of 15.3 years (range of 1 to 41 years) compared to controls (Fawer et al. 1983). To estimate an equivalent continuous exposure concentration, the average concentration assumed for an 8-hour exposure was adjusted to 24-hours, 7 days/week, as shown above for the USEPA RfC. Uncertainty factors of 10 for protection of sensitive subpopulations and 3 for use of a LOAEL instead of a NOAEL were applied to 0.0062 mg m^{-3} (the adjusted 24-hour exposure concentration) to yield a chronic inhalation MRL of 0.0002 mg m^{-3}.

Other Thresholds for Toxicological Effects

In the mid-1980s, Albers et al. (1988) investigated evidence of neurological abnormalities in workers at the Y-12 Plant, including workers who were exposed to airborne elemental mercury between 1953 and 1963. Subjects included individuals who worked at the plant and were exposed to mercury for at least four months during this period. During the exposure period, mercury workers at Y-12 underwent at least quarterly measurement of urine mercury levels. In addition, air samples were collected. The cohort was

rank ordered based on cumulative exposure between 1953-1986 (defined as the sum of average quarterly urinary mercury in mg L^{-1}). A questionnaire was completed detailing lifestyle habits including smoking, caffeine consumption, and alcohol consumption, as well as exposure to other neurotoxicants at the Y-12 Plant and other occupational and nonoccupational sources. Several measures of mercury exposure were considered to evaluate evidence of dose-response relationships, including cumulative exposure, length of exposure, and peak exposure. The study did not identify how measured urine levels related to air concentrations. Effects of mercury exposure were evaluated using clinical examination of such measures as strength, coordination, and reflexes, and quantitative measurements of grip-strength, pressure sensations, and nerve conduction.

Overall, few significant differences between the exposed and control group were observed, and these differences were very small. In the clinical and quantitative examinations the only significant group difference observed was a significantly increased quantitative tremor in the exposed group. Positive correlations between declining neurological performance and increasing mercury exposure were observed using multiple linear regression analysis of cumulative and peak urine mercury with neurological examination measures. These results suggest a weak but significant dose-response relationship between abnormal neurological function and increasing mercury exposure, although the study results presented are insufficient to quantify the relationship between exposure levels and response.

Limited data on developmental or reproductive effects of inhaled elemental mercury in pregnant women or in laboratory animals suggest that reproductive or developmental effects following exposure to high concentrations of elemental mercury vapor may be a concern. For example, Mishinova et al. (1980) reported that the rates of pregnancy and labor complication (reproductive effects) were higher among women exposed to elemental mercury vapors in the workplace than unexposed workers, though dose-response relationships were not established. A study by Steffek et al. (1987) showed an increase in the number of fetal resorptions in pregnant rats exposed to 0.5 mg m^{-3} of elemental mercury for 20 days, but not at 0.1 mg m^{-3}. An animal study evaluated the developmental effects of elemental mercury exposure of two week old male rats on behavior at 4 to 6 months of age. Exposure to 0.05 mg m^{-3} for one or four hours per day for one week resulted in subtle behavior changes (including decreases in spatial learning) (Fredriksson et al. 1992). In two subsequent studies (Danielsson et al. 1993; Frederiksson et al. 1996), pregnant rats were exposed to air concentrations of 1.8 mg m^{-3} for one to three hours per day for six to eight days; behavioral changes were noted in the offspring . In an assessment of male fertility, Lauwerys et al. (1985) found no significant decrease in the observed number of children born to an exposed group (estimated mean air concentration 0.043 mg m^{-3}) compared to the controls. All of the exposure levels associated with developmental effects were at air concentrations higher than those associated with observations of CNS effects in adult workers (0.023 to 0.033 mg m^{-3}). Therefore, toxicity benchmarks based on these lower concentrations are likely to be protective of developmental or reproductive effects.

Selection of Toxicity Benchmarks for Comparison to Estimated Doses

The relevant data from the studies discussed above are summarized in Table 11-2. As discussed above, the USEPA RfD and the ATSDR MRL for inhalation of elemental mercury range from 0.000057 to 0.000086 mg kg^{-1} d^{-1}**S** these values are both based on studies in humans. LOAELs from chronic duration studies in humans range from 0.017 to 0.076 mg m^{-3}, while NOAELs range from 0.025 to 0.1 mg m^{-3}. The preponderance of data suggest that a NOAEL of approximately 0.010 mg m^{-3} (calculated by dividing an average LOAEL of 0.030 mg m^{-3} by a LOAEL-to-NOAEL adjustment factor of 3) to 0.025 mg m^{-3} is appropriate as an upper-bound toxicity benchmark for evaluating chronic exposures to elemental mercury vapors. These NOAELs, based on neurological effects in humans, are expected to be lower than NOAELs for other toxicological endpoints, including developmental and reproductive effects.

11.4 Summary of Dose-Response Data for Methylmercury

Perhaps the largest body of data and most vigorous ongoing investigations of mercury toxicity involve exposures to methylmercury. Methylmercury may be found in the environment as the result of direct releases from industrial processes. In addition, methylmercury is created in the environment from methylation of inorganic mercury by microorganisms. A unique factor that contributes to the potential for methylmercury exposure is that it is concentrated through the aquatic food chain, such that virtually all fish contain some methylmercury as the result of bioconcentration of natural and anthropogenic (from human sources) methylmercury. As discussed in Section 5 of this report, for purposes of evaluating exposures to mercury in fish caught near the ORR, estimated doses from consumption of fish were compared to threshold doses based on methylmercury exposures.

Studies of toxicological effects following exposure to methylmercury, and selection of toxicity benchmarks for comparison to estimated doses, are described below.

Toxicokinetics of Ingested Methylmercury

Studies have shown that methylmercury ingested through fish consumption is readily absorbed through the GI tract (~95%) and that about 5% of absorbed methylmercury partitions to the blood (ATSDR 1994). Methylmercury can cross the blood-brain barrier and the placenta; thus, women can pass methylmercury to the fetus during pregnancy.

The critical effect associated with exposures to methylmercury is assumed to be neurological effects on the developing fetus. Developmental toxicity in infants exposed to methylmercury *in utero* (prior to birth) has been manifested in various levels of brain damage (ranging from cerebral palsy, seizures, and mental retardation to learning disabilities), abnormal muscle tone or reflexes, and delayed onset of walking and talking. Neurological effects associated with post-natal exposures (following birth) have been observed following short term exposures to high levels of methylmercury in fish. Effects observed following post-natal exposures included paresthesia (tingling sensation in the extremities), tremors, abnormal reflexes, speech difficulties, and impaired peripheral vision.

Table 11-2: Summary of Dose-Response Data for Exposure to Elemental Mercury Vapor

Study	Type of Exposure	Mean Exposure Duration (y)	Measured Mean Air Hg Conc. ($mg\ m^{-3}$)	Mean Blood or Urine Hg Conc. [a]	Measure of Hg Toxicity	Threshold Air Conc. ($mg\ m^{-3}$)
Albers et al. (1988)	Worker exposure, Y-12 Plant	N/A	N/A	N/A	Neurological performance, incl. tremor	N/A
Buchet et al. (1980)	Worker exposure, chlor-alkali plant	6.8 y	N/A	Hg-B = 28.5 µg L^{-1} Hg-U = 59.1 µg Hg g^{-1} creatinine (~52.6 µg L^{-1})	Kidney function	0.048 (LOAEL) [b]
Ehrenberg et al. (1991)	Worker exposure, thermometer manuf. facility	N/A	0.076 (TWA, personal samplers)	Hg-U= 73.2 µg Hg g^{-1} creatinine (~65.1 µg L^{-1})	Questionnaire; neurological exam; renal function	0.076 (LOAEL) [c]
Fawer et al. (1983)	Worker exposure, various manufacturing facilities	15.3 y (range 1 - 41 y)	0.026 (TWA, personal samplers)	Hg-B = 41.3 µmol L^{-1} Hg-U = 11.3 µmol mol^{-1} (20.1 µg L^{-1})	Hand tremor	0.026 (LOAEL) [c]
Langworth et al. (1992)	Workers, chlor-alkali plant	13.5 y	0.025 (avg; peaks up to 0.15 µg/m3)	Hg-B (median) = 11.0 µg L^{-1} Hg-U (median) = 25.4 µg Hg g^{-1} creatinine (~22.6 µg L^{-1})	Neurobehaviorial function	0.025 (LOAEL)[c]
Lauwerys et al. (1983)	Worker exposure, chlor-alkali plant or amalgam factory	5.5 y	N/A	Hg-B = 16.5 µg L^{-1} Hg-U = 53.1 µg Hg g^{-1} creatinine (~47.3 µg L^{-1})	Kidney and immune function	0.044 (NOAEL) [b]
Lauwerys et al. (1985)	Worker exposure, chemical or manufacturing plant	8.7 y	N/A	Hg-B = 14.6 µg L^{-1} Hg-U = 52.4 µg Hg g^{-1} creatinine (~46.6 µg L^{-1})	Male fertility	0.043 (NOAEL) [b]
Levine et al. (1982)	Worker exposure, chlor-alkali plant	N/A	N/A	Hg-U = 290 µg L^{-1}	Neurotoxicity (nerve conduction tests)	N/A

Study	Type of Exposure	Mean Exposure Duration (y)	Measured Mean Air Hg Conc. ($mg\ m^{-3}$)	Mean Blood or Urine Hg Conc. [a]	Measure of Hg Toxicity	Threshold Air Conc. ($mg\ m^{-3}$)
Liang et al. (1993)	Worker exposure, lamp factory	10.4 y	0.033	Hg-U = 25 µg L^{-1}	Neurobehaviorial function	0.033 (LOAEL)[c]
Ngim et al. (1992)	Worker exposure, dentists	5.5 y	0.014 (TWA, personal samplers)	Hg-B = 12.3 µg L^{-1}	Neurobehaviorial function	0.023 (LOAEL)[c]
Piikivi (1989)	Worker exposure, chlor-alkali plant	16 y	N/A	Hg-B = 11.8 µg L^{-1} Hg-U = 19.3 µg L^{-1}	Cardiovascular reflexes	0.030 (LOAEL)[b]
Piikivi and Hanninen (1989)	Worker exposure, chlor-alkali plant	13.7 y	N/A	Hg-B = 10 µg L^{-1} Hg-U = 17 µg L^{-1}	Neurobehaviorial function	0.025 (LOAEL)[b]
Piikivi and Ruokonen (1989)	Worker exposure, chlor-alkali plant	13.7 y	N/A	Hg-B =10.3 µg L^{-1}	Kidney function (urine proteins)	0.025 (NOAEL) [b]
Piikivi and Tolonen (1989)	Worker exposure, chlor-alkali plant	15.6 y	N/A	Hg-B = 11.6 µg L^{-1} Hg-U = 11.6 µmol mol^{-1} creat. (20.6 µg L^{-1})	Cerebral effects (electroencephalography (EEG))	0.025 (LOAEL) [b]
Rosenman et al. (1986)	Worker exposure, chemical plant	appx. 4-5 y (median)	N/A	Hg-B (median) = ~28-50 µg L^{-1}	Neurobehaviorial and kidney function	0.050 (LOAEL)[b]
Smith et al. (1970)	Worker exposure, chlor-alkali plants	6-9 y (median)	0-0.27 (measured TWA)	N/A	CNS effects (e.g., tremor, insomnia)	0.1 (NOAEL) [c]
Stonard et al. (1983)	Worker exposure, chlor-alkali plant	8 y (range 1 - 33 y)	N/A	Hg-U = 67 µg Hg g^{-1} creatinine (~60 µg L^{-1})	Renal function (urine proteins)	0.055 (NOAEL) [b]
Verberk et al. (1986)	Worker exposure, fluorescent lamp factory	5 y (median, range 0.5 - 19 y)	N/A	Hg-U = 20 µmol mol^{-1} creatinine (35.5 ug L^{-1})	Postural tremor of the finger	0.017 (LOAEL)[b]

a Hg-B = Mercury concentration in blood; Hg-U = Mercury concentration in urine
b Air concentration estimated based on Roels; i.e., 1:1.22 (air (µg/m3):urine (µg/g creatinine))
c Air concentration measured
N/A Not available

Dose-Response Data for Evaluating Methylmercury Exposures

The body of dose-response data for methylmercury is comprised primarily of three groups of exposure data:

- Exposures to mercury in fish from Minamata Bay in 1953-1960 and Niigata, Japan in 1965, contaminated by releases of methylmercury from chemical manufacturing plants;

- Exposures of an Iraqi population to methylmercury-treated seed grain in home-baked bread in 1971-72; and

- Exposures of fish-eating populations to lower levels of methylmercury in fish, including studies in the Seychelles Islands, northern Quebec, and New Zealand.

The Minimata Bay and Niigata methylmercury exposures provided the first indications that the fetal brain is the most sensitive target organ for methylmercury exposures. Studies conducted using data collected from populations exposed to methylmercury in contaminated seed grain in Iraq resulted in the development of the first dose-response relationships for methylmercury exposures. The RfD for exposure to methylmercury developed by the USEPA is based on data from the Iraqi exposures. Questions have been raised, however, about the applicability of dose-response relationships based on this exposure incident for predicting the potential of adverse health effects from ingestion of methylmercury in fish. For example, studies have suggested that if selenium is present and bioconcentrates in fish along with methylmercury, the selenium may protect against methylmercury toxicity due to selenium's antioxidant properties (Grandjean et al. 1992). Because of these concerns, a number of studies have been initiated to investigate the dose-response relationship associated with ingestion of methylmercury in fish.

Most studies investigating dose-response relationships for methylmercury exposures focus on *in utero* exposures as the most sensitive exposure group. The majority of these studies establish dose -response relationships based on maternal blood concentrations extrapolated from mercury concentrations measured in hair. Concentrations in blood during pregnancy can be estimated even after an exposure has ended because mercury concentrations remain unchanged once incorporated into the hair shaft, and concentrations of mercury in newly formed hair are proportional to the simultaneous concentration in the blood. By assuming that hair growth occurs at a rate of 1 cm per month, concentrations in blood in previous months can be calculated.

Typically, blood concentrations are estimated assuming a constant hair:blood concentration ratio. For example, the USEPA assumes a hair:blood concentration ratio of 250: 1 (i.e., 250 $\mu g\ kg^{-1}$ hair: 1 $\mu g\ L^{-1}$ blood), based on a range of reported ratios between 140:1 and 370:1 (IRIS 1998). Blood concentrations can then correlated to daily dietary intake of methylmercury (in $\mu g\ d^{-1}$) as follows:

$$\textit{Daily dietary intake}\ (\mu g\ d^{-1}) = \frac{C_{blood} \times b \times V}{A \times f} \qquad (11.2)$$

where:

C_{blood}	=	Concentration in blood (µg L^{-1})
b	=	Elimination constant (d^{-1})
V	=	Volume of blood in the body (L)
A	=	Fraction of mercury in the diet (i.e., fish) that is absorbed (unitless)
f	=	Fraction of daily intake taken up by blood (unitless)

Dose-Response Relationships for Methylmercury Based on Iraqi Data

The Iraqi exposures to methylmercury occurred in 1971 and 1972 when a large population consumed home-baked bread that had been prepared using methylmercury-treated seed grain. The seed grain had been treated with methylmercury because of methylmercury's fungicidal properties. Exposures to many individuals were very high. The worst health effects were noted in infants born to mothers who had consumed the methylmercury while pregnant. After the exposures ceased, several studies of the women and their offspring were conducted to evaluate the incidence of adverse neurological effects in the children, and to determine if a dose-response relationship could be established (Bakir et al. 1973; Clarkson et al. 1976; Cox et al. 1989; Marsh et al. 1980; Marsh et al. 1981; Marsh et al. 1987). Evidence of adverse neurological effects was based on age of walking and talking, muscle tone, tendon reflexes, and other indicators.

The USEPA used data from the Iraqi exposure incident to develop an RfD for ingestion of methylmercury, to protect infants from developmental neurologic abnormalities. The RfD of 1×10^{-4} mg kg^{-1} d^{-1} is based on dose-response data presented by Marsh et al. (1987) and the Food and Drug Administration (FDA) (Seafood Safety 1991). Marsh et al. (1987) estimated maternal blood levels of mercury that infants were exposed to in the womb by measuring the mercury content of maternal hair segments and relating these concentrations to blood levels. The FDA, in turn, conducted a risk assessment using the Marsh et al. (1987) data, dividing the 81 mother-infant pairs into five dose groups to facilitate use of a benchmark dose approach. Using these data, the USEPA determined the dose associated with an excess (above background) risk of a combination of adverse childhood neurologic effects, including delayed onset of walking and talking, low neurologic scores, mental symptoms, and seizures. The USEPA chose as a dose-response threshold the dose at which a benchmark response (BMR) of 10% of the children showed delayed development. The lower bound of the 95% confidence range of the dose corresponding to the BMR (11 ppm mercury in maternal hair) was used as the basis for the RfD. USEPA assumes that the 95% lower confidence limit of a 10% response roughly correlates with a NOAEL for fetal developmental toxicity data (Faustman et al. 1994; Allen et al. 1994a,b).

The maternal hair concentration was converted to a daily dose assuming a hair:blood concentration ratio of 250: 1, an elimination constant (*b*) of 0.014 d^{-1}, a blood volume (*V*) of 5.4 L, an absorption fraction (*A*) of 0.95, and a fraction of daily intake taken up by blood (*f*) of 0.05. The resulting daily dose was 70 µg d^{-1}, or 0.001 mg kg^{-1} d^{-1}, assuming an average body weight of 60 kg. The RfD was derived by incorporating a total uncertainty factor of 10 (3 for lack of a two-generation reproductive study and lack of data on the effect of exposure duration, and 3 for variation in the half-life of methylmercury and variation in the hair:blood ratio).

Dose-Response Relationships for Methylmercury Based on Fish Consumption Data

Several studies of populations that consume large amounts of mercury-contaminated fish have been conducted or are on-going to evaluate the dose-response relationship for ingestion of methylmercury in fish. These include studies of several populations in different parts of the world whose diets consist largely of ocean fish, including studies in the Seychelles Islands, northern Quebec, New Zealand, and the Faroe Islands. Data from these studies are described below.

Seychelles Islands

The goal of the ongoing Seychelles Child Development Study is to establish whether there is a relationship between fetal exposure to methylmercury from dietary fish, at lower exposure levels than in the epidemic poisonings in Iraq and Japan, and measures of developmental effects in early childhood. The study is being conducted using data gathered from a population in the Republic of Seychelles, an island nation in the Indian Ocean, that consumes large amounts of ocean-caught fish. One advantage of the Seychelles location is the absence of mercury from local industrial emissions— the islands are 1,000 miles from any continent or large population center and mercury exposures are associated only with consumption of ocean fish (median number of fish meals consumed per week is 12).

A pilot study was initiated in 1987 to provide guidance for the main study, which began in 1989. The study population in the main study consisted of a cohort of 789 children, and was conducted as a double-blind study using maternal hair mercury as the index of fetal exposure. Maternal hair mercury levels ranged from 0.5-26.7 ppm with a median of 5.9 ppm. Each child was evaluated at 19 months and again at 29 months for infant intelligence (Bayley Scales of Infant Development) and Mental and Psychomotor Scales, with a modified version of the BSID Infant Behavior Record to measure adaptive behaviors at 29 months.

An association between fetal mercury exposure and developmental endpoints was found in the pilot study but not in the main study, and only when questionable test scores were treated as negative results in the analysis. The effect disappeared when the questionable results were treated as positive results (Myers et al., 1995). The main study found no effect that could be attributed to mercury on the BSID scores obtained at either the 19-month or the 29-month interval. There was one subjective observation of a slight decrease in activity level determined by the BSID Infant Behavior Record in boys (but not girls) that might be attributable to prenatal mercury exposure (Marsh et al., 1995; Myers et al., 1995; ATSDR 1997).

However, according to the investigators, it is still unclear if an association between low-level fetal mercury exposure and neurodevelopmental effects exists.

Northern Quebec

A group of 247 children in four communities of Cree Indians in northern Quebec was evaluated to determine if a dose-reponse releationship could be established between methylmercury exposure from consumption of locally-caught fish and neurodevelopmental effects. Children first exhibited clinical signs consistent with methylmercury exposure (at 6 -20 ppm maternal hair mercury) between 12 and 30 months of age. Delayed deep tendon reflexes in the legs were positively associated with methylmercury exposure in boys, but there was no consistent dose-response relationship. The study was confounded by evidence of alcoholism and smoking among mothers (McKeown-Eyssen et al. 1983).

New Zealand

Thirty-one matched pairs of mothers and infants from populations of different ethnic backgrounds in New Zealand were selected based on consumption of at least three fish meals per week during pregnancy. Retrospective mercury concentrations in maternal hair were determined for the gestation period (mean 9 ppm maternal hair mercury for the high-exposure group and 2 ppm for the reference group), and mental development of the children was tested at four and six years of age. Although delays in fine motor and language areas were noted in some children at four years of age, differences between the experimental and control groups were not significant. Analysis of results of neurodevelopmental tests conducted at six years of age suggested that the psychological test variables were influenced by ethnic background and social class. An association between prenatal methylmercury exposure and decreased test performance was found, but only contributed a small part of the variation in test results significance (Kjellstrom al. 1989). A recent reanalysis of the data suggested that positive associations between methylmercury exposure and decreased performance in the tests, found in only one of three cultural groups, may be due to the presence of some confounding variable in that cultural group (Crump et al. 1996).

Faroe Islands

In 1986, a large study was initiated in fishing communities in the Faroe Islands to evaluate neurodevelopmental effects of methylmercury and PCB exposure *in utero* (Grandjean et al. 1997). Increased mercury exposure in the population is largely attributed to the eating of pilot whale, which is traditionally hunted and shared among individuals in the communities (ATSDR 1997). A total of 917 mother-child pairs were evaluated; mercury was measured in maternal hair and cord blood, and children were evaluated for neurodevelopmental effects at about seven years of age. The median maternal hair mercury concentration was 4.5 ppm, and 13% had hair concentrations greater than 10 ppm. Three neurological tests were found to be difficult for 7-year old children, with fewer than 60% of the children performing optimally. With inclusion of covariates with uncertain influence on the test results, multiple regression analysis indicated that nine out of 20 measures showed mercury related detriments. The authors

concluded that adverse effects are observed at exposures below 10 ppm in maternal hair (Grandjean et al. 1997).

ATSDR MRL for Ingestion of Methylmercury

ATSDR developed a chronic MRL for oral exposure to methylmercury of 0.0005 mg kg^{-1} d^{-1}, based on data from the Seychelles Child Development Study (ATSDR 1997). The MRL was established based on the relationship between concentrations of mercury measured in maternal hair during pregnancy and decrements in neurobehavioral test performance by children at 29 months of age. The mothers of these children had been exposed to methylmercury through fish ingestion before and during pregnancy and throughout lactation (Davidson et al. 1995). The 29-month cohort of 736 children represented 94% of the 779 pairs initially enrolled in the study and approximately 50% of all live births in the Seychelles Islands in 1989. This particular study population was selected to establish the MRL because: (1) they regularly consumed a high quantity and variety of ocean fish; (2) pre-study mercury concentrations in maternal hair were in the appropriate range (5-45 ppm) to evaluate low-level mercury exposures; (3) there is no local industry that emits environmental pollutants, and the Seychelles location is 1,000 mi. from any continent or large population center; (4) the population is highly literate, cooperative, and has minimal immigration; and (5) the population is generally healthy, with low maternal alcohol and tobacco use (ATSDR 1997).

Maternal hair concentrations measured in hair segments corresponding to the time of pregnancy ranged from 0.5 to 26.7 ppm, with a median exposure of 5.9 ppm for the entire study group. As discussed above, there was one subjective observation of a slight decrease in activity level as determined by the BSID Infant Behavior Record in boys (but not girls) that might be attributable to prenatal mercury exposure. However, the effect on activity level in boys is not considered an adverse effect and the 5.9 ppm level is categorized as a NOAEL (ATSDR 1997).

The median maternal hair concentration of 5.9 ppm was converted to a daily dose of 29 µg d^{-1}, assuming a hair:blood concentration of 250:1, an absorption fraction for mercury in the diet of 0.95, a fraction of absorbed dose found in the blood of 0.05, an elimination constant of 0.014 d^{-1}, a blood volume of 4.2 L (female), and a body weight of 60 kg (female). The estimated dose is 0.0005 mg kg^{-1} d^{-1}. No uncertainty factors were needed since the MRL was derived from a NOAEL and the exposure group was made up of mother-infant pairs, the most appropriate and most sensitive population for this end point.

USEPA Mercury Study Report to Congress Recommended Criteria for Ingestion of Methylmercury

The USEPA Mercury Study Report to Congress, prepared by the USEPA and submitted to Congress as required under the Clean Air Act, provides an assessment of the magnitude of U.S. mercury emissions by source, the health and environmental implications of those emissions, and the availability and cost of control technologies (USEPA 1997). In *Volume V: Health Effects of Mercury and Mercury Compounds*, the current USEPA RfD of 0.0001 mg kg^{-1} d^{-1} is used to determine the magnitude of adverse health effects from U.S. mercury emissions associated with ingestion of methylmercury in fish. A Science Advisory Board report is quoted as saying, "The current RfD, based on the Iraqi and New Zealand data, should be

retained at least until the on-going Faeroe and Seychelles Islands studies have progressed much further and been subjected to the same scrutiny as has the Iraqi data. The RfD may need to be reassessed in terms of the most sensitive endpoints from these new studies."

Selection of Toxicity Benchmarks for Comparison to Estimated Doses

The relevant data from the studies discussed above are summarized in Table 11-3. As discussed above, the USEPA RfD and the ATSDR MRL for ingestion of methylmercury range from 0.0001 to 0.0005 mg kg^{-1} d^{-1}**S** these values are both based on studies in humans. These data suggest that a NOAEL for exposure to methylmercury of 0.0005 mg kg^{-1} d^{-1} is appropriate as an upper-bound toxicity benchmark for evaluating *in utero* exposures to methylmercury from maternal consumption of fish.

Table 11-3: Summary of Dose-Response Data for Exposure to Methylmercury

Study	Type of Exposure	Maternal Hair Concentration	Measure of Hg Toxicity	Threshold Dose (mg kg^{-1} d^{-1})
Iraqi data (Marsh et al. 1987)	MeHg in seed grain	11 ppm (threshold)	Neurological test scores in children	0.001 (LOAEL)
Seychelles (Marsh et al. 1995; Myers et al. 1995)	MeHg in fish	5.9 ppm (median)	Neurological test scores in children	0.0005 (NOAEL)
Faroe Islands (Grandjean et al. 1997)	MeHg in fish	4.5 ppm (median); 10 ppm (87^{th} %ile)	Neurological test scores in children	Not estimated; neurological effects at hair conc. < 10 ppm

11.5 Toxicity Benchmarks for Comparison with Task 2 Results

Table 11-4 summarizes toxicity benchmarks for the key mercury species and exposure routes evaluated in this assessment. When comparing these toxicity benchmarks to estimated doses, it is important to remember the basis for these values. As discussed above, NOAELs represent the highest dose in a given study at which no statistically or biologically significant indication of the toxic effect of concern has been identified. Thus, doses *below* these levels were not associated with toxic effects in the population being studied. However, these reported NOAELs may be based on animal studies, or they may be based on human exposure populations that are different in a number of key parameters, such as age- or health-status or duration of exposure, from the exposed populations evaluated in the dose reconstruction. Additionally, as presented, these NOAELs do not incorporate factors to account for minimal data or uncertainties in extrapolating from animal data to threshold doses in humans.

USEPA RfDs and ATSDR MRLs are set to reflect exposure levels below which adverse health effects in exposed populations, *including* sensitive subpopulations, are not expected to occur. It is generally assumed that doses below these levels are not likely to be associated with adverse health risks, and therefore are less likely to be of regulatory concern, and that as the frequency and/or the magnitude of the doses exceeding these levels increases, the probability of adverse effects in a human population increases. It should not be categorically concluded, however, that all doses below the USEPA or ATSDR levels, or any other regulatory criteria, are "acceptable" (or will be risk-free) and that all doses in excess of these levels are "unacceptable" (or could result in adverse health effects) (Barnes and Dourson 1988).

All of the studies discussed in Section 11.4 for evaluating methylmercury exposures are based on *in utero* exposures, since the fetus is considered to be particularly sensitive to methylmercury exposure. However, observations of CNS effects in children exposed *in utero* were associated with no or slight transient symptoms in the mother, even at the high concentrations associated with the Minamata, Japan, and Iraq exposures (Clarkson et al. 1985; Marsh et al. 1987 Clarkson 1990). Therefore, a different threshold is considered appropriate for evaluating adult exposures. Both the FDA action level of 1 ppm for methylmercury in fish and a previous (1985) USEPA RfD of 0.0003 mg kg^{-1} d^{-1} are based on neurological effects in adults— these values will be used for comparison to doses estimated in the dose reconstruction for adults.

Table 11-4: Toxicity Benchmarks for Comparison with Results of the Mercury Dose Reconstruction

Species and Exposure Route	No Observed Adverse Effect Level ($mg\ kg^{-1}\ d^{-1}$)	USEPA Reference Dose[a] ($mg\ kg^{-1}\ d^{-1}$)	ATSDR Minimal Risk Level[b] ($mg\ kg^{-1}\ d^{-1}$)
Ingestion of Inorganic Mercury	0.1 to 0.23 (animal studies)[c]	0.0003	0.002[d]
Inhalation of Elemental Mercury	0.0029 to 0.0071 (human studies)[e]	0.000086[f]	0.000057[g]
Ingestion of Methylmercury— *In utero* and child exposure	0.0005 (human studies)[h]	0.0001	0.0005[i]
Ingestion of Methylmercury— Adult exposure	NA	0.0003[j]	NA

NOTES:

a Reference: IRIS 1998.

b Reference: ATSDR 1997.

c Minimal data are available. Both data points given are from studies in laboratory animals. The lower limit of the range is based on the intermediate duration LOAEL used as the basis for the USEPA RfD, divided by an adjustment factor of 3 to extrapolate from a LOAEL to a NOAEL. The upper limit is based on the NOAEL from the NTP intermediate duration study.

d For intermediate duration exposures.

e Both values given are from studies in humans. The lower limit of the range is based on a NOAEL of 0.010 $mg\ m^{-3}$ (calculated by dividing an average LOAEL of 0.030 $mg\ m^{-3}$ by a LOAEL-to-NOAEL adjustment factor of 3). The upper limit is based on a NOAEL of 0.025 $mg\ m^{-3}$. $Mg\ kg^{-1}\ d^{-1}$ doses are calculated by multiplying the NOAELs by a breathing rate of 20 $m^3\ d^{-1}$ and dividing by a body weight of 70 kg.

f Derived by multiplying USEPA's reference concentration (3×10^{-4} $mg\ m^{-3}$) by a breathing rate of 20 $m^3\ d^{-1}$ and dividing by a body weight of 70 kg.

g Derived by multiplying ATSDR's Minimal Risk Level for chronic exposure (2×10^{-4} $mg\ m^{-3}$) by a breathing rate of 20 $m^3\ d^{-1}$ and dividing by a body weight of 70 kg.

h The data point given is based on studies in humans. Based on ATSDR's estimated 29 Fg d^{-1} dose necessary to achieve 5.9 ppm in maternal hair, which was the NOAEL associated with the Seychelles Child Development Study. The 29 Fg d^{-1} dose was converted to 0.029 $mg\ d^{-1}$ and then divided by a body weight of 60 kg to yield a NOAEL of 0.0005 $mg\ kg^{-1}\ d^{-1}$.

i For chronic exposure.

J Reference: USEPA 1985.

NA Not available.

12.0 EVALUATION OF THE POTENTIAL FOR HEALTH EFFECTS IN EXPOSED POPULATIONS

This section compares mercury doses estimated for each population to the toxicity benchmark values presented in Section 11. The purpose of this section is to put the dose estimates in perspective and discuss the likelihood that the estimated levels of historical exposure could have resulted in adverse health effects in exposed populations.

12.1 Comparison of Estimated Doses with Toxicity Benchmark Values

The following sections describe the annual average doses of mercury estimated for each of the evaluated populations. For each population, figures are presented that show the 95% subjective confidence interval on the estimated annual average daily mercury doses for 1950-1990. Toxicity benchmark values corresponding to applicable USEPA reference doses (RfDs) and no observable adverse effect levels (NOAELs) are also indicated. In addition, for each population, summary tables outline the years that the 95% subjective confidence intervals on estimated doses exceeded the RfD or NOAEL. When the *upper bound* (97.5^{th} percentile) on the estimated doses was *below* the RfD, it is not likely that adverse health effects occurred in that population as a result of exposures to mercury from the Y-12 Plant during that year, based on current scientific knowledge. Exceeding the RfD is equivalent to exceeding a *hazard index* of 1 (hazard indices are calculated by dividing the estimated dose by the appropriate RfD).

More detailed results summaries are presented in Appendix X. For each population, Table X-1 presents the estimated elemental, total inorganic, and methylmercury doses at the 97.5^{th} percentile (upper confidence limit or "UCL"), the 50^{th} percentile ("Central" estimate), and the 2.5^{th} percentile (lower confidence limit or "LCL") for each year. Elemental mercury doses were assumed to be from the inhalation of airborne mercury pathway; methylmercury doses were assumed to be from the fish consumption pathway; and total inorganic mercury pathways were assumed to be the sum of doses from all of the remaining pathways. Doses equal to or greater than the RfD are shaded in Table X-1. Table X-2 presents the hazard indices corresponding to each dose presented in Table X-1. Hazard indices equal to or greater than 1.0 are shaded (indicating that the dose is equal to or greater than the RfD).

12.1.1 Wolf Valley Residents, 1953-1962

The Wolf Valley Resident population reflects exposures to individuals assumed to live down valley from the Y-12 Plant in the Wolf Valley area on the opposite side of the Clinch River. These individuals were assumed to be exposed to mercury through inhalation and ingestion of "backyard" meat, milk, and fruits/vegetables contaminated by airborne mercury. Figures 12-1 and 12-2 show estimated elemental mercury doses (from inhalation) for this population for 1953-1962. Figures 12-3 and 12-4 show estimated total inorganic mercury doses for all of the inorganic mercury pathways combined. The highest doses to this population were estimated to have been in 1955, since this was the year that mercury releases to air from Y-12 were estimated to be greatest.

Figure 12-1: Wolf Valley Residents-
Comparison of Estimated Elemental Mercury Doses (Adult)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

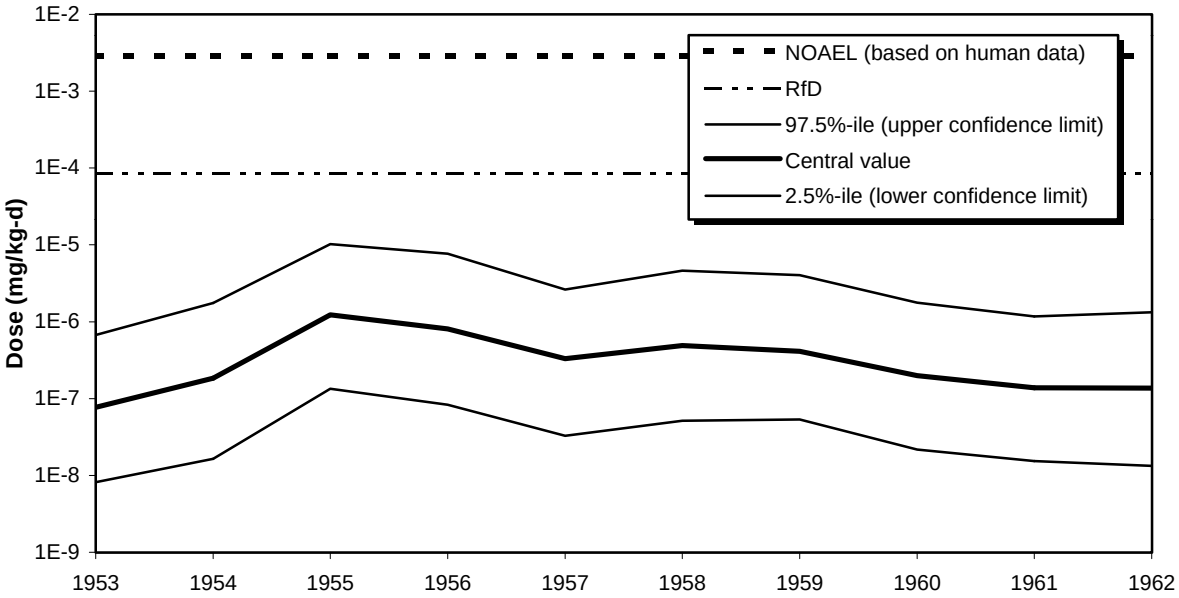


Figure 12-2: Wolf Valley Residents-
Comparison of Estimated Elemental Mercury Doses (Child)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

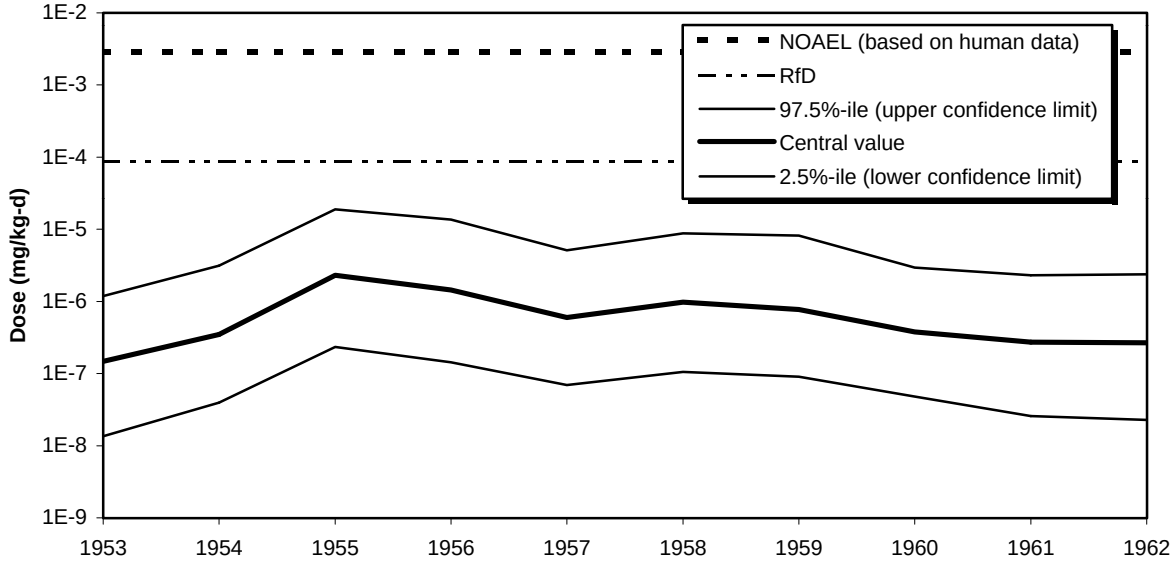


Figure 12-3: Wolf Valley Residents-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Adult)
with Toxicity Benchmark Values for Inorganic Mercury

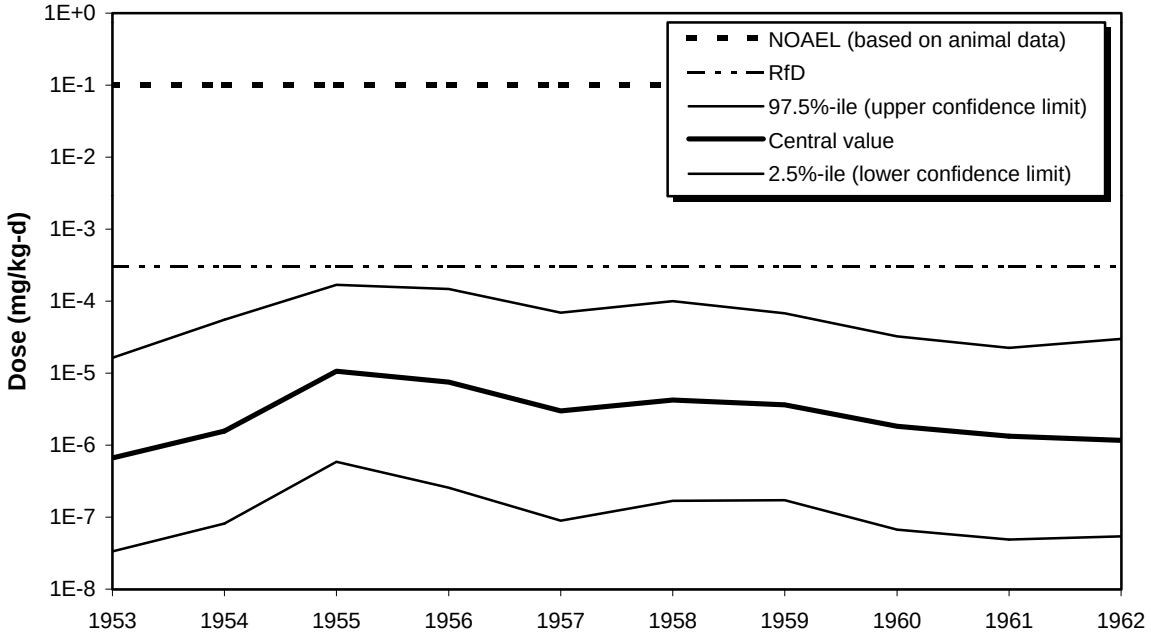


Figure 12-4: Wolf Valley Residents-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Child)
with Toxicity Benchmark Values for Inorganic Mercury

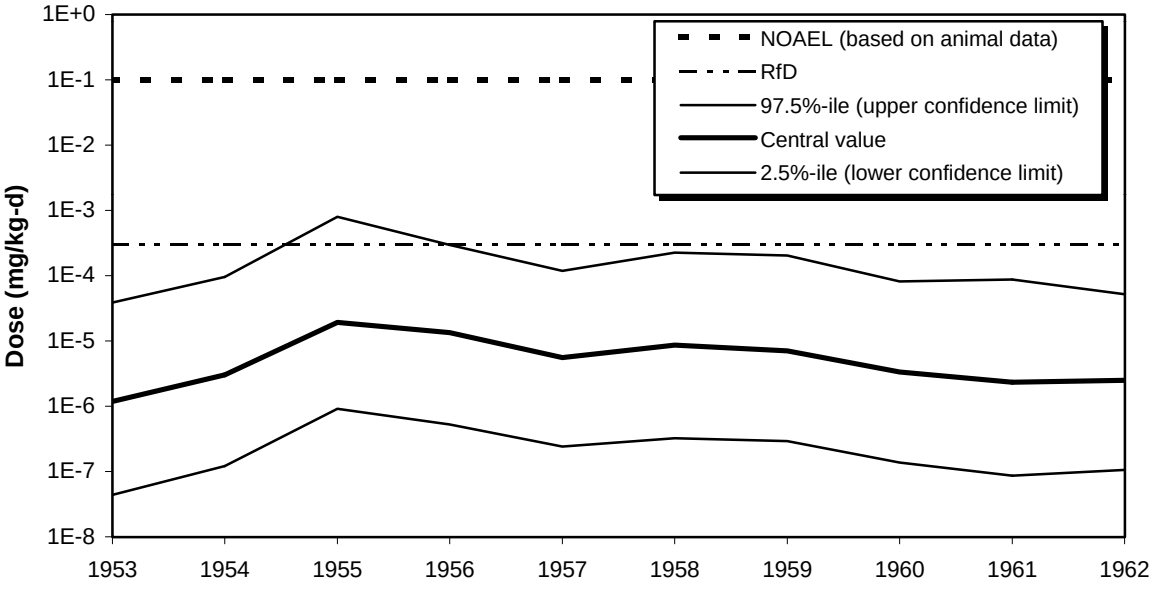


Table 12-1 summarizes the years that the estimated annual average elemental and inorganic mercury doses exceeded the RfD or the NOAEL, and presents the ratios of the highest estimated doses to the RfDs and NOAELs. Ratios in excess of 1.0 indicate that the RfD or NOAEL was exceeded.

Table 12-1: Wolf Valley ResidentS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	Estimated Population Size [a]	%-ile	Years Dose Exceeded the:		Ratio of Highest Dose (at given %-ile) to:	
			USEPA RfD [b]	NOAEL [c]	USEPA RfD [b]	NOAEL [c]
Adult						
Elemental mercury	15-50	97.5%ile	---	---	0.12 *(1955)*	0.0034 *(1955)*
		50%ile	---	---	0.014 *(1955)*	0.00041 *(1955)*
		2.5%ile	---	---	0.0015 *(1955)*	0.000045 *(1955)*
Inorganic mercury	15-50	97.5%ile	---	---	0.57 *(1955)*	0.0017 *(1955)*
		50%ile	---	---	0.037 *(1955)*	0.00011 *(1955)*
		2.5%ile	---	---	0.0020 *(1955)*	0.0000059 *(1955)*
Child (6 mo - 3 yrs)						
Elemental mercury	5-15	97.5%ile	---	---	0.22 *(1955)*	0.0066 *(1955)*
		50%ile	---	---	0.027 *(1955)*	0.00079 *(1955)*
		2.5%ile	---	---	0.0027 *(1955)*	0.000079 *(1955)*
Inorganic mercury	5-15	97.5%ile	1955	---	**2.7** *(1955)*	0.0080 *(1955)*
		50%ile	---	---	0.0063 *(1955)*	0.00019 *(1955)*
		2.5%ile	---	---	0.0031 *(1955)*	0.0000092 *(1955)*

a Annual average population size
b USEPA RfD for elemental mercury = 0.000086 mg kg^{-1} d^{-1}; USEPA RfD for inorganic mercury = 0.00030 mg kg^{-1} d^{-1}
c NOAEL for elemental mercury = 0.0029 mg kg^{-1} d^{-1}; NOAEL for inorganic mercury = 0.10 mg kg^{-1} d^{-1}

Estimated elemental mercury doses were below the USEPA RfDs for all years evaluated. The *upper bound* on the child's total inorganic mercury dose for 1955 was slightly above the inorganic mercury RfD, but less than 1/100 of the NOAEL. While inorganic mercury doses slightly above the NOAEL have been associated with kidney effects in laboratory rodents, health effects in humans exposed to inorganic mercury at doses at or below the NOAEL have not been reported.

As shown in Figure 10-4, the primary contributor to the estimated total inorganic mercury doses to Wolf Valley Residents was ingestion of homegrown fruits and vegetables contaminated by airborne mercury (contributing approximately 99% of the mean total inorganic mercury doses estimated for 1955). Other pathways (ingestion of milk and ingestion of beef) contributed only about 1% or less of the estimated total inorganic mercury dose. Sensitivity analyses show that the dominant contributor to variance in estimated doses through ingestion of homegrown fruits/vegetables is the ingestion rate. Parameters also contributing significantly to variance are the air concentration and factors describing deposition of airborne mercury to plant surfaces (detailed results of the sensitivity analysis are summarized in Appendix Y).

The variance in the homegrown above-ground fruit/vegetable rate distribution was assumed to be largely a factor of interindividual variability, rather than uncertainty. In other words, because of individual differences in eating habits and the availability of homegrown fruits/ vegetables for consumption, a range of consumption rates was assumed for adult and child members of this population. The upper bound of the ingestion rate distribution for children between 6 months and 3 years of age was assumed to be slightly more than one-half pound of fresh homegrown fruits/vegetables per day. The central value was assumed to be about two-tenths of a pound per day (fresh weight). Therefore, it is unlikely that a child would have been exposed to mercury at doses exceeding the USEPA RfD unless he/she consumed relatively large amounts of homegrown fruits and vegetables. For adults, the upper bound of the ingestion rate distribution was assumed to be about 1.3 pounds per day (fresh weight), and the central value was assumed to be about 0.44 pounds per day (fresh weight). Table 12-2 shows the 95% subjective confidence interval on the amount of homegrown fruits/vegetables that a child or adult member of the Wolf Valley Residents population would have had to consume in a given year to be exposed to inorganic mercury at an annual average dose equal to the USEPA RfD.

Table 12-2: 95% Subjective Confidence Intervals on the Pounds of Homegrown Above-Ground Fruits/ Vegetables (fresh weight) Consumed per Year that Would Have Yielded an Inorganic Mercury Dose Equal to the USEPA RfDS Wolf Valley Resident [a]

Year	Child (6 mo - 3 yrs)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1953	160	1,900	>2,000	1,100	>2,000	>2,000
1954	61	850	>2,000	240	>2,000	>2,000
1955	11	130	1,300	65	620	>2,000
1956	14	180	>2,000	71	990	>2,000
1957	38	420	>2,000	210	>2,000	>2,000
1958	25	290	>2,000	140	1,500	>2,000
1959	28	310	>2,000	180	1,800	>2,000
1960	59	730	>2,000	320	380	>2,000
1961	85	1,000	>2,000	450	>2,000	>2,000
1962	71	1,000	>2,000	550	>2,000	>2,000

a Estimated consumption rates greater than 2,000 pounds per year (i.e., about 5.5 pounds per day, fresh weight) are considered highly unlikely, and are indicated as "> 2,000".

b The 95%ile confidence intervals for adults were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution). 50th %ile fruit/vegetable consumption rates for adult males equal to the RfD would likely be about 25% higher due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

As shown, during the year of highest mercury releases to air (1955), a child who consumed 11 or more pounds of homegrown *above-ground* fruits/vegetables (i.e., excluding potatoes, onion, or other "root" crops) per year may have been exposed to mercury in contaminated fruits/vegetables at doses that exceeded the USEPA inorganic mercury RfD. If, during that year, the child consumed more than 130 pounds of homegrown above-ground fruits/vegetables, it is *likely* that the child's dose exceeded the RfD.

Population Size

As discussed in Section 6.2, the estimated total size of the Wolf Valley Resident population during a given year was assumed to be 30 to 100 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 15 to 50 individuals), adult females of child-bearing age comprised approximately 20% of the population (or about 6 to 20 individuals) and children ages 6 months to 3 years comprised approximately 15% of the population (or about 5 to 15 individuals).

Conclusions

Based on this analysis, it is unlikely that Wolf Valley Residents were exposed to mercury from Y-12 at levels that exceeded the USEPA RfDs, unless they consumed very large quantities of homegrown above-ground fruits and vegetables. Because of the small size of this population and the relatively low doses estimated for them, it is likely that the number of individuals in this population who were exposed to mercury at doses above the RfD, if any, was small.

12.1.2 Scarboro Community Residents, 1950-1990

The Scarboro Community Residents population reflects exposures to individuals who lived in the Scarboro Community on the opposite side of Pine Ridge from the Y-12 Plant. These individuals were assumed to be exposed to mercury through inhalation; dermal contact with soil and EFPC sediment and water; and ingestion of soil, EFPC sediment and water, homegrown fruits/vegetables, and EFPC fish. Figures 12-5 and 12-6 show estimated elemental mercury doses (from inhalation) for Scarboro Community adults and children for 1950-1990. Figures 12-7 and 12-8 show estimated total inorganic mercury doses for all of the inorganic mercury pathways combined (the remaining pathways with the exclusion of fish consumption). Figures 12-9 and 12-10 show estimated methylmercury doses from consumption of EFPC fish. The highest doses were estimated to have been in 1955, since this was the year that mercury releases to air and water from Y-12 were estimated to be greatest.

Table 12-3 summarizes the years that the estimated annual average elemental, inorganic, and methyl mercury doses exceeded the USEPA RfD or the NOAELs, and presents the ratios of the highest estimated doses to the RfDs and NOAELs. Ratios in excess of 1.0 indicate that the RfD or NOAEL was exceeded.

Exposures to Elemental Mercury

For the adult, estimated elemental mercury doses (from inhalation) were below the USEPA RfD for all years evaluated. For the child, *upper bounds* on the estimated elemental mercury doses exceeded the RfD for three years: 1955, 1957, and 1958. All doses were less than 1/24 of the NOAEL.

Elemental mercury doses slightly above the NOAEL have been associated with neurological effects in adult workers exposed to airborne mercury**S** the neurological effects observed included hand tremor, increases in memory disturbances, and evidence of dysfunction of the autonomic (involuntary) nervous system (IRIS 1998). At slightly higher doses, evidence of effects on the kidney have also been observed. However, health effects in humans exposed to elemental mercury at doses at or below the NOAEL have not been reported.

Figure 12-5: Scarboro Community Residents-
Comparison of Estimated Elemental Mercury Doses (Adult)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

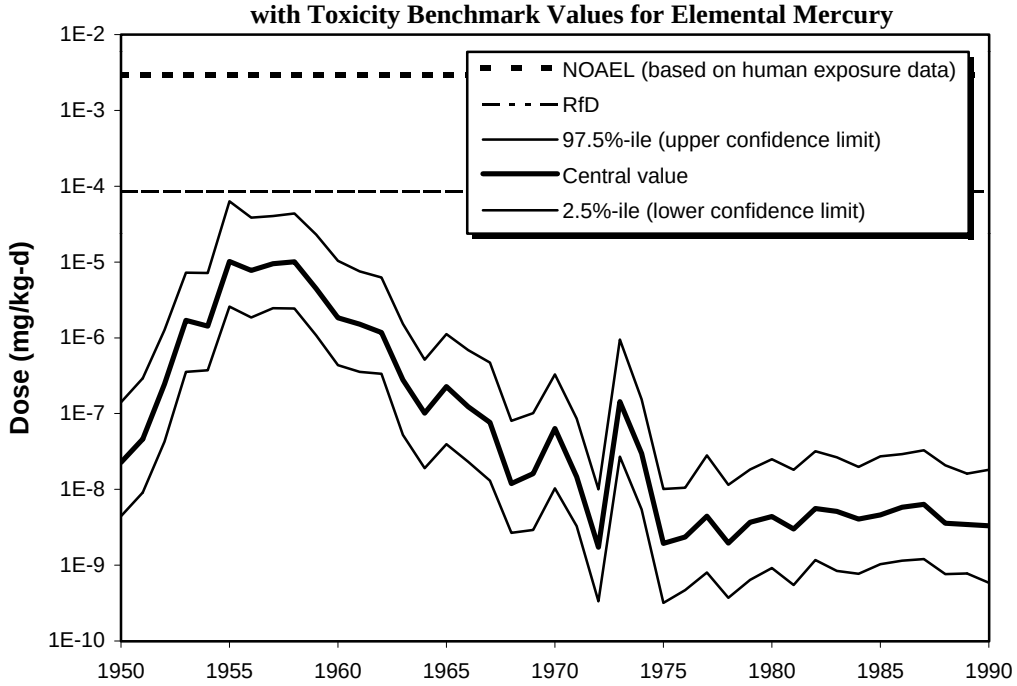


Figure 12-6: Scarboro Community Residents-
Comparison of Estimated Elemental Mercury Doses (Child)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

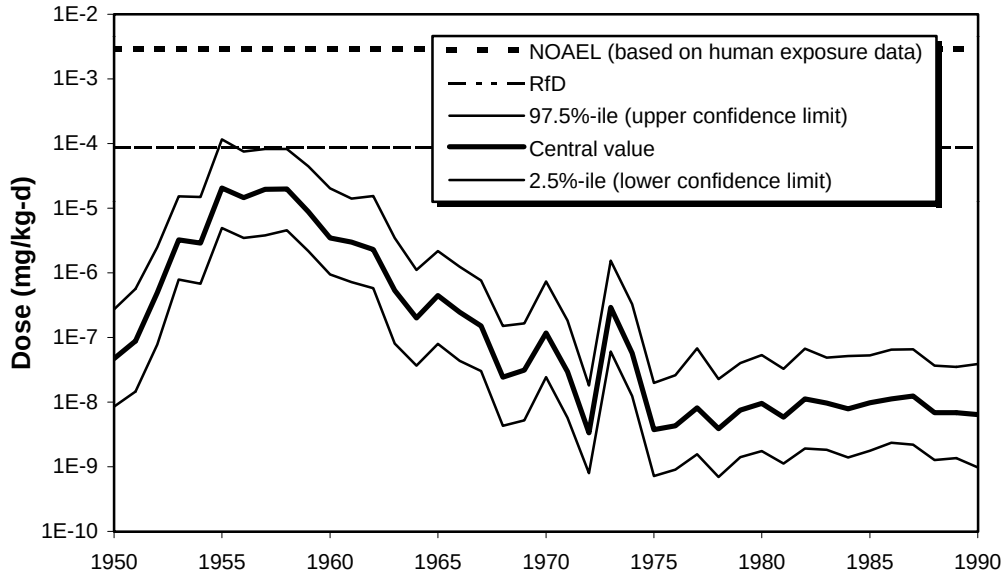


Figure 12-7: Scarboro Community Residents-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Adult)
with Toxicity Benchmark Values for Inorganic Mercury

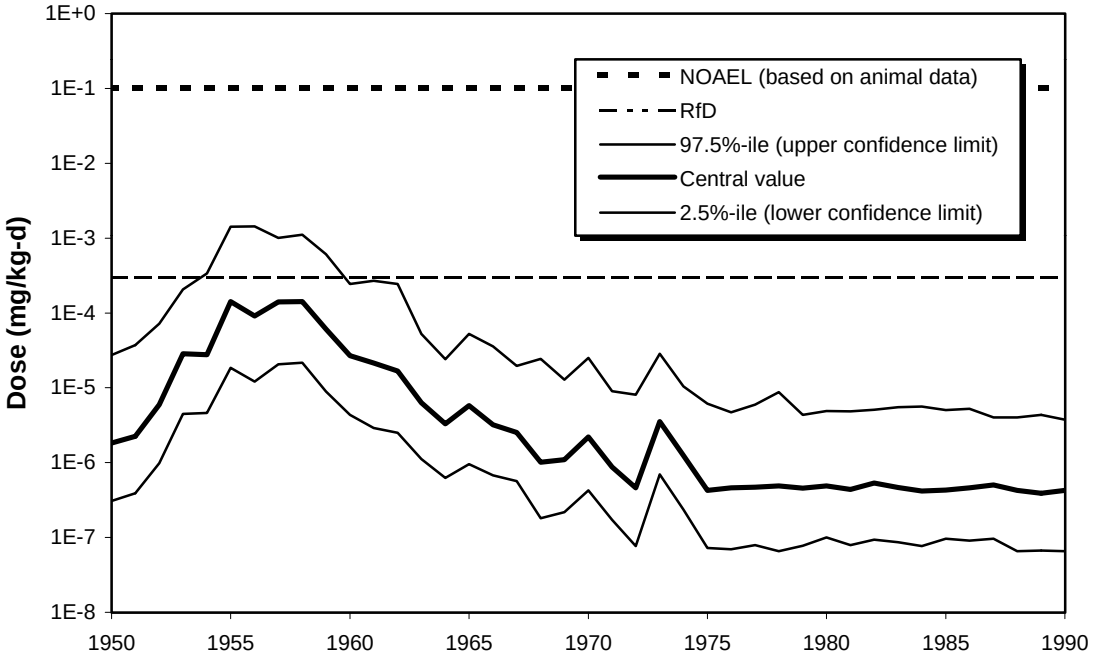


Figure 12-8: Scarboro Community Residents-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Child)
with Toxicity Benchmark Values for Inorganic Mercury

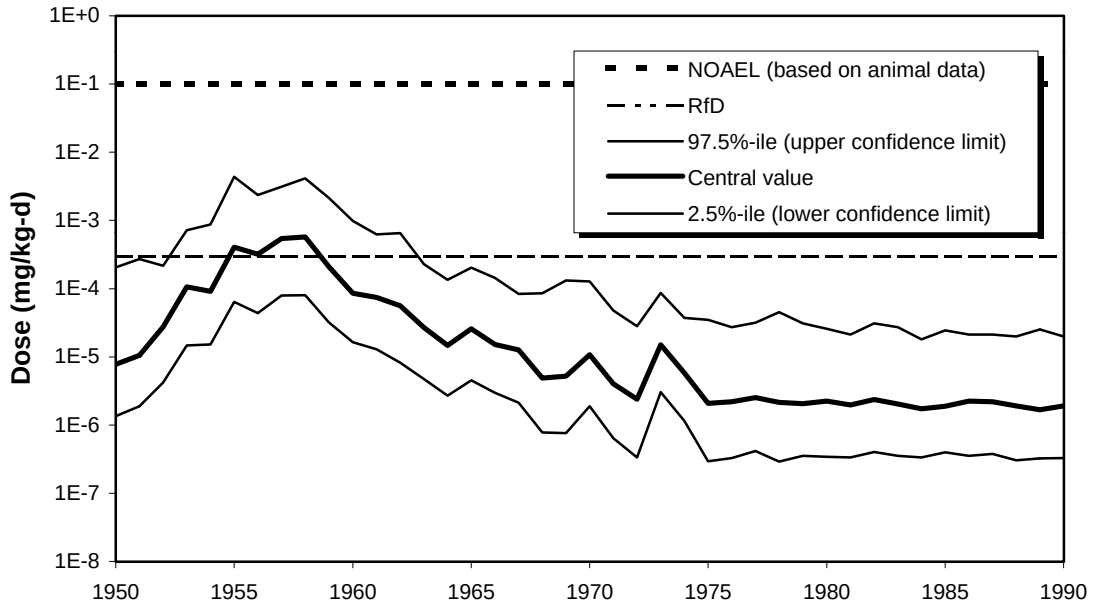


Figure 12-9: Scarboro Community Residents-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

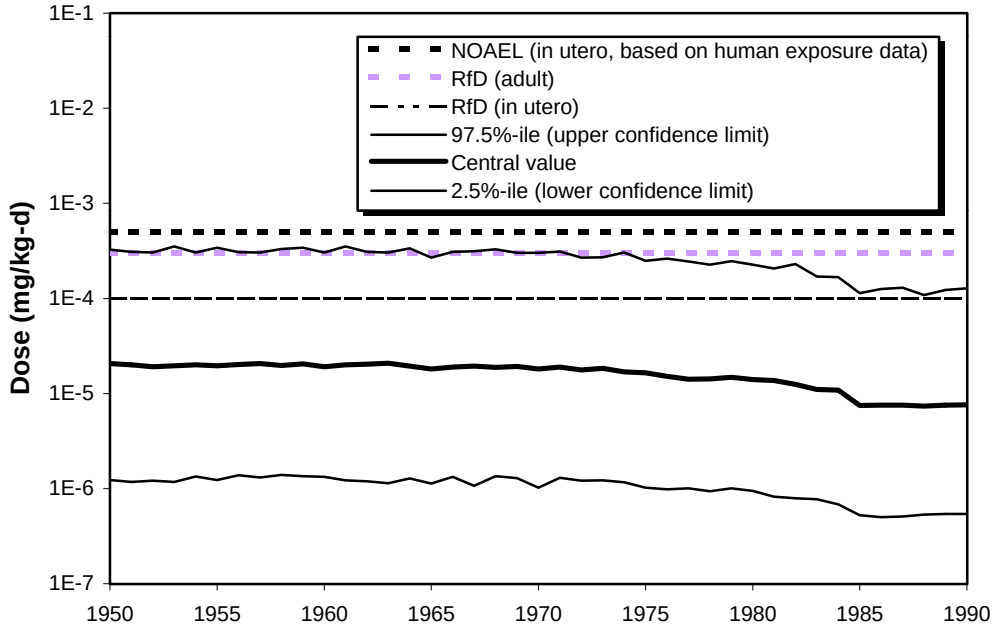


Figure 12-10: Scarboro Community Residents-
Comparison of Estimated Methylmercury Doses (Child)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

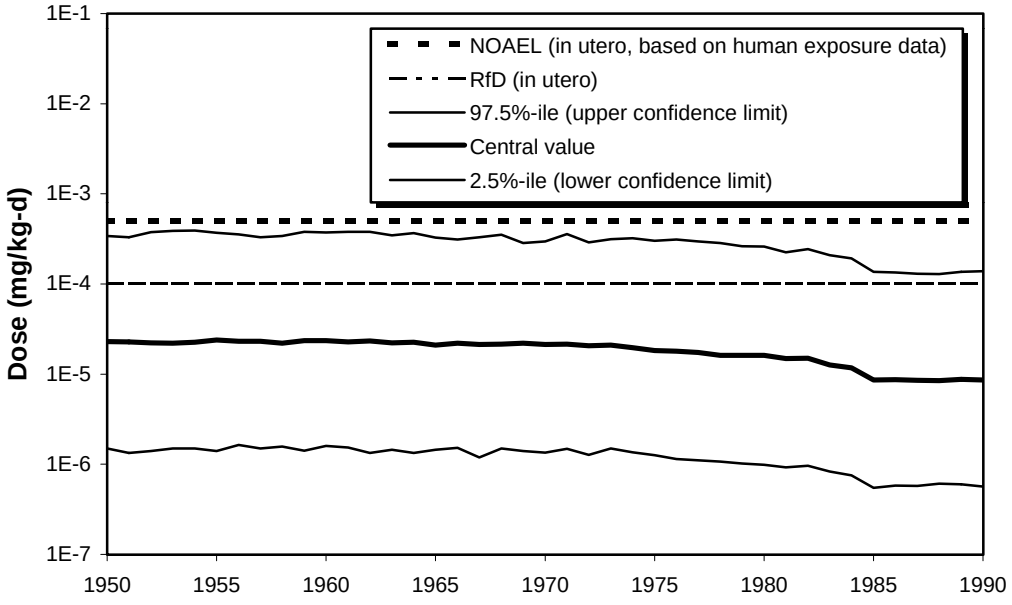


Table 12-3: Scarboro Community ResidentS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	Estimated Population Size [a]	%-ile	Years Dose Exceeded the:		Ratio of Highest Dose (at given %-ile) to:	
			USEPA RfD [b]	NOAEL [c]	USEPA RfD [b]	NOAEL [c]
Adult						
Elemental mercury	400-600	97.5%ile	---	---	0.67 *(1955)*	0.020 *(1955)*
		50%ile	---	---	0.13 *(1955)*	0.0038 *(1955)*
		2.5%ile	---	---	0.033 *(1955)*	0.00097 *(1955)*
Inorganic mercury	400-600	97.5%ile	1954-59	---	**4.7** *(1955)*	0.014 *(1955)*
		50%ile	---	---	0.47 *(1955)*	0.0014 *(1955)*
		2.5%ile	---	---	0.063 *(1955)*	0.00019 *(1955)*
Methyl-mercury	160-240, 400-600	97.5%ile	1950-90 [1954-73] [d]	---	**3.6 [1.2]** [d] *(1957)*	0.72 *(1957)*
		50%ile	---	---	0.21 [0.07] [d] *(1955)*	0.042 *(1955)*
		2.5%ile	---	---	0.013 [0.0043] [d] *(1957)*	0.0026 *(1957)*
Child (6 mo - 3 yrs)						
Elemental mercury	120-180	97.5%ile	1955, 1957-58	---	**1.4** *(1955)*	0.041 *(1955)*
		50%ile	---	---	0.26 *(1955)*	0.0076 *(1955)*
		2.5%ile	---	---	0.058 *(1955)*	0.0017 *(1955)*
Inorganic mercury	120-180	97.5%ile	1953-62	---	**15** *(1955)*	0.044 *(1955)*
		50%ile	1955-58	---	**1.4** *(1955)*	0.0041 *(1955)*
		2.5%ile	---	---	0.21 *(1955)*	0.00064 *(1955)*
Methyl-mercury	120-180	97.5%ile	1950-90	---	**3.9** *(1956)*	0.78 *(1956)*
		50%ile	---	---	0.24 *(1956)*	0.048 *(1956)*
		2.5%ile	---	---	0.014 *(1956)*	0.0028 *(1956)*

a Annual average; for methylmercury adult exposure, the first range is for women of child-bearing age and the second range is for all adults

b USEPA RfD for elemental mercury = 0.000086 mg kg^{-1} d^{-1}; USEPA RfD for inorganic mercury = 0.00030 mg kg^{-1} d^{-1}; USEPA RfD for methylmercury (*in utero* exposure)= 0.00010 mg kg^{-1} d^{-1}; RfD for methylmercury (adult exposure) = 0.00030 mg kg^{-1} d^{-1}

c NOAEL for elemental mercury = 0.0029 mg kg^{-1} d^{-1}; NOAEL for inorganic mercury = 0.10 mg kg^{-1} d^{-1}; NOAEL for methylmercury (*in utero* exposure) = 0.0005 mg kg^{-1} d^{-1}

d First value represents comparison to *in utero* exposure RfD, second value represents comparison to adult exposure RfD

Sensitivity analyses showed that the dominant contributor to the variance in the inhalation dose estimates was the air concentration term. As discussed in Section 7.2.2 and shown in Figure 7-2, airborne mercury concentrations at Scarboro during 1953 to 1962 were assumed to be contributed by two sources: direct airborne mercury releases from Y-12 that were transported over Pine Ridge, and volatilization of mercury from EFPC surface water to air. During these years, air concentrations at Scarboro (from both sources) were estimated to be between 15% and 40% of the air concentrations estimated for the EFPC Floodplain Farm Family location. The higher estimated air concentrations at the EFPC Floodplain Farm Family location were due to its closer proximity to EFPC. Although about 25% to 60% of the total airborne mercury at Scarboro during these years were estimated to come from volatilization of mercury from EFPC, the air concentration at Scarboro from volatilization *only* was estimated to be only about 10% of the air concentration estimated at the EFPC Floodplain Farm Family location (which was assumed to come entirely from volatilization from EFPC).

Air concentration estimates from Y-12 and EFPC were associated with a significant degree of uncertainty because of the lack of any direct measurements of airborne mercury in Scarboro, and uncertainties about how much mercury could have been transported over Pine Ridge and how much volatilized from EFPC. These uncertainties could not be resolved within the constraints of this project; consequently, wide uncertainty bounds were placed on the parameters describing airborne concentrations of mercury in the Scarboro community.

Exposures to Inorganic Mercury

Estimated total inorganic mercury doses (from all inorganic mercury pathways combined) exceeded the USEPA inorganic mercury RfD for the following years:

- *Adults***S** *Upper bounds* on the estimated total inorganic mercury doses exceeded the RfD for 1954-1959. Doses for these years were less than 1/70 of the NOAEL.

- *Children***S** *Upper bounds* on the estimated total inorganic mercury doses exceeded the RfD for 1953-1962; *central values* exceeded the RfD for 1955-1958. Doses for these years were less than 1/20 of the NOAEL.

Inorganic mercury doses slightly above the NOAEL have been associated with kidney effects in laboratory rodents. Health effects in humans exposed to inorganic mercury at doses at or below the NOAEL have not been reported. The RfD is approximately 1/300 of the NOAEL**S** the large difference between the RfD and the NOAEL is the result of safety factors incorporated into the RfD to account for insufficient data on the toxicity of inorganic mercury to humans at low dose/ long term exposures.

As shown in Figure 10-4, the pathway with the greatest contribution to the estimated total inorganic mercury doses for Scarboro residents was ingestion of homegrown above-ground fruits and vegetables contaminated by airborne mercury. In 1955, 92% of the mean total inorganic mercury dose to adults was

estimated to be contributed by this pathway; for children, the contribution was approximately 78%. In 1958, approximately 84% of the mean adult dose was estimated to be contributed by this pathway; for children, the contribution was approximately 54%. Other pathways contributing significantly to the child's estimated total inorganic mercury dose were incidental ingestion of EFPC surface water and dermal contact with EFPC surface water (for 1955-1958, estimated *upper bound* inorganic mercury doses from these pathways were at or slightly above the USEPA inorganic mercury RfD).

Sensitivity analyses show that the dominant contributor to variance in estimated doses through ingestion of fruits/vegetables contaminated by airborne mercury is the ingestion rate. Parameters also contributing significantly to variance are air concentration and factors describing deposition of airborne mercury to plant surfaces (detailed results of the sensitivity analysis are summarized in Appendix Y). For the contact with EFPC surface water pathways, the dominant contributors to variance are the surface water ingestion rate, duration and frequency of exposure, and the surface area of exposed skin.

Table 12-4 shows the 95% subjective confidence interval on the amount of homegrown above-ground fruits/vegetables that a child or adult would have had to consume in a given year to be exposed to inorganic mercury at an annual average dose (through this pathway) equal to the USEPA RfD. As shown, during the years of highest mercury releases from Y-12 (1953 to 1962), children or adults consuming even moderate amounts of homegrown above-ground fruits or vegetables may have been exposed to mercury at doses exceeding the USEPA RfD for mercury. For example, if a child consumed approximately 1-1/2 pounds (fresh weight) of homegrown above-ground fruits/ vegetables during 1955, there is a small probability that the child's inorganic mercury dose was equal to or greater than the RfD. If a child consumed about 12 or more pounds (fresh weight) of homegrown above-ground fruits/vegetables during that year (an average of about 1/3 pound per day), it is likely that the child's inorganic mercury dose was equal to or greater than the USEPA RfD. For most years (i.e., before 1953 and after 1962) , it is unlikely that Scarboro residents would have been exposed to mercury through the homegrown fruit/vegetable pathway at doses exceeding the USEPA RfD, because it is unlikely that an individual would have consumed sufficiently large amounts of homegrown fruits and vegetables.

Table 12-5 shows the 95% subjective confidence interval on the number of hours per year that a child or adult would have had to have been exposed to EFPC surface water (assuming that exposure occurred through both incidental ingestion and dermal contact) to have received an annual average inorganic mercury dose (through these pathways) equal to the USEPA RfD. As shown, for most years, a child would have had to spend a very large number of hours in the creek to have been exposed to inorganic mercury at doses exceeding the RfD. During the years of highest mercury releases to EFPC (1957 and 1958), a child who spent about 8 or more hours per year in the creek may have been exposed to mercury at doses exceeding the RfD. If the child spent 18 or more hours in the creek during these years, it is likely that the child was exposed to mercury at doses exceeding the RfD. During 1953-1956 and 1959-1962, children who played in the creek more frequently (approximately 15 to 50 or more hours per year), may have been exposed to mercury at doses exceeding the RfD.

Table 12-4: 95% Subjective Confidence Intervals on the Pounds of Homegrown Above-Ground Fruits/Vegetables (fresh weight) Consumed per Year that Would Have Yielded an Inorganic Mercury Dose Equal to the USEPA RfDS Scarboro Resident [a]

Year	Child (6 mo - 3 yrs)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	480	>2,000	>2,000	>2,000	>2,000	>2,000
1951	330	>2,000	>2,000	>2,000	>2,000	>2,000
1952	60	650	>2,000	400	>2,000	>2,000
1953	11	74	660	50	400	>2,000
1954	8.2	71	490	48	380	>2,000
1955	1.5	12	120	7.3	64	580
1956	2.2	16	180	11	89	720
1957	2.0	15	120	11	77	660
1958	2.2	13	130	11	70	610
1959	3.8	29	270	19	150	1,200
1960	7.7	64	540	37	330	>2,000
1961	8.8	81	970	56	460	>2,000
1962	13	100	1,000	59	570	>2,000
1963	44	460	>2,000	300	>2,000	>2,000
1964	180	1,400	>2,000	810	>2,000	>2,000
1965	57	700	>2,000	410	>2,000	>2,000
1966	150	1,200	>2,000	740	>2,000	>2,000
1967	220	16	>2,000	1,200	>2,000	>2,000
1968	1,400	>2,000	>2,000	>2,000	>2,000	>2,000
1969	950	>2,000	>2,000	>2,000	>2,000	>2,000
1970	290	>2,000	>2,000	1,600	>2,000	>2,000
1971	990	>2,000	>2,000	>2,000	>2,000	>2,000
1972	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1973	110	830	>2,000	560	>2,000	>2,000
1974	560	>2,000	>2,000	>2,000	>2,000	>2,000
>1974 (per year)	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000

a Estimated consumption rates greater than 2,000 pounds per year (i.e., about 5.5 pounds per day, fresh weight) are considered highly unlikely, and are indicated as "> 2,000".

b The 95%ile confidence intervals for adults were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution). 50th %ile fruit/vegetable consumption rates for adult males equal to the RfD would likely be about 25% higher due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

Table 12-5: 95% Subjective Confidence Intervals on the Hours of Exposure to EFPC Surface Water per Year that Would Have Yielded an Inorganic Mercury Dose Equal to the USEPA RfDS Scarboro Community Resident [a]

Year	Child (6 mo - 3 yrs)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	2,000	5,700	>8,760	7,300	>8,760	>8,760
1951	1,000	2,800	>8,760	3,600	>8,760	>8,760
1952	210	580	2,400	710	2,400	>8,760
1953	45	130	600	170	530	2,300
1954	71	200	970	280	870	3,300
1955	15	43	210	62	170	770
1956	20	52	270	70	210	910
1957	8.7	19	75	32	79	310
1958	7.8	18	68	30	71	240
1959	27	59	260	110	250	800
1960	77	180	690	310	740	2,900
1961	89	210	800	330	850	3,200
1962	140	370	1,500	550	1,400	5,800
1963	190	480	2,600	750	2,000	>8,760
1964	390	1,000	4,600	1,600	4,000	>8,760
1965	190	460	2,100	670	1,900	7,000
1966	400	990	4,900	1,500	4,100	>8,760
1967	550	1,500	6,800	2,200	6,200	>8,760
1968	3,600	8,200	>8,760	>8,760	>8,760	>8,760
1969	3,100	6,900	>8,760	>8,760	>8,760	>8,760
1970	730	1,600	7,000	2,700	6,600	>8,760
1971	3,100	7,100	>8,760	>8,760	>8,760	>8,760
1972	>8,760	>8,760	>8,760	>8,760	>8,760	>8,760
1973	290	700	2,800	1,200	2,800	>8,760
1974	1,300	2,900	>8,760	5,000	>8,760	>8,760
>1974 (per year)	>6,000	>8,760	>8,760	>8,760	>8,760	>8,760

a Estimated exposure times that exceed the number of hours in a year are indicated as ">8,760"

b The 95%ile confidence intervals for adults were calculated using body weights and skin surface areas representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution, std dev = 9.5) and an average skin surface area of 8000 cm^2 (lognormal distribution, std dev = 800). 50^{th} %ile hours for adult males equal to the RfD would likely be about 10-15% higher due to their higher average body weight and skin surface area (the average body weight of an adult male is about 78 kg and the average skin surface area is about 10,000 cm^2, USEPA 1995).

Tables 12-4 and 12-5 present 95% subjective confidence intervals of ingestion rates or exposure times that would have resulted in doses that exceeded the RfD if each pathway was the *only* pathway through which the individual was exposed. If an individual consumed homegrown fruits/vegetables *and* played in EFPC, both pathways would contribute to the total inorganic mercury dose, and lower fruit/vegetable ingestion rates or exposure times to EFPC water than shown in Tables 12-4 and 12-5 could have resulted in total inorganic mercury doses from both sources combined that exceeded the RfD.

Exposures to Methylmercury

Methylmercury doses to adult females from consumption of fish were compared to two different measures of potential toxicity— the USEPA methylmercury RfD, which is intended to be protective of neurological effects in children that are exposed *in utero*, and a separately derived RfD intended to be protective of adverse health effects in adults. Methylmercury doses that have been associated with adverse health effects in children exposed *in utero* (during the Minimata, Japan or Iraq poisoning episodes) have not been associated with adverse health effects in the exposed mother, and it is assumed that adults are less sensitive to adverse health effects from methylmercury exposure than the unborn child.

The *upper bound* (97.5^{th} percentile) of the estimated methylmercury doses to adults and children from consumption of fish caught in EFPC exceeded the USEPA methylmercury RfD (based on *in utero* exposures) for all years evaluated (1950-1990). However, for all years, estimated adult and child doses were less than the NOAEL that is the basis for the RfD. Methylmercury doses slightly above the NOAEL have been associated with observations of neurological effects in children who were exposed to methylmercury *in utero* (i.e., their mothers consumed methylmercury in fish). This RfD is set at a dose level one-fifth of the NOAEL. The *upper bound* (97.5^{th} percentile) of the estimated methylmercury doses to adult females exceeded the adult RfD for 1954-1973.

As indicated in Section 6.2.1.6, interviews with Oak Ridge area residents, including residents of the Scarboro Community, suggest that the maximum rate of consumption of fish from EFPC was about one fish meal per month (DaMassa 1995). In this assessment, the average rate of consumption of fish from EFPC was assumed to be 1.2 g d^{-1} (or about 2.5 meals per year assuming 170 g per meal), and the upper bound (95^{th} percentile) was assumed to be about 4.6 g d^{-1} (or about 10 meals per year assuming 170 g per meal).

Table 12-6 shows the 95% subjective confidence interval on the number of meals of fish from EFPC that a child would have had to consume per year to receive an annual average methylmercury dose equal to the USEPA RfD for methylmercury (based on *in utero* exposure, assumed to be protective of health effects in developing young children). Table 12-7 shows the 95% subjective confidence interval on the number of meals of fish from EFPC an adult female would have had to consume per year to receive annual average methylmercury doses equal to the RfDs for methylmercury based on *in utero* exposures and based on the adult RfD. For purposes of this evaluation, the size of a fish meal for a child was assumed to range between 50 g and 120 g, and the size of a fish meal for an adult female was assumed to range between 80 g and 250 g (a filet of fish about the size of a deck of cards weighs approximately 100 g).

Table 12-6: 95% Subjective Confidence Intervals on the Meals of Fish from EFPC Consumed per Year that Would Have Yielded a Methylmercury Dose Equal to the RfD (based on *in utero* exposure) [a, b]

Year	Child exposure (compared to *in utero* RfD)		
	2.5%ile	50%ile	97.5%ile
1950	1.0	2.0	4.1
1951	0.9	1.9	4.0
1952	1.0	1.9	3.9
1953	0.9	1.9	4.0
1954	0.9	1.9	4.4
1955	1.0	1.9	4.3
1956	0.9	1.9	3.9
1957	1.0	1.9	4.1
1958	1.0	1.9	4.1
1959	1.0	2.0	4.4
1960	0.9	2.0	4.1
1961	1.0	1.9	3.9
1962	0.9	1.9	4.0
1963	0.9	2.0	4.0
1964	0.9	1.9	3.9
1965	1.0	2.0	4.0
1966	1.1	2.0	4.0
1967	1.0	2.1	4.0
1968	1.1	2.0	4.2
1969	1.1	2.0	3.9
1970	1.0	2.1	4.5
1971	1.0	2.1	4.3
1972	1.1	2.1	4.1
1973	1.1	2.2	4.1
1974	1.1	2.1	4.9
1975	1.2	2.3	4.5
1976	1.2	2.4	5.0
1977	1.3	2.5	5.2
1978	1.3	2.6	5.0
1979	1.4	2.7	5.3
1980	1.4	2.8	5.8
1981	1.5	3.0	5.7
1982	1.6	3.0	5.7
1983	1.7	3.4	6.5
1984	2.0	3.7	7.4
1985	2.7	5.0	10.3
1986	2.8	5.1	9.1
1987	2.7	5.0	9.1
1988	3.0	5.0	9.7
1989	2.7	5.0	9.7
1990	2.9	5.2	9.6

a Child fish meal consumption rates equal to the RfD are based on the USEPA methylmercury RfD for *in utero* exposure of 0.0001 mg kg^{-1} d^{-1}.

b The 95%ile confidence intervals were calculated using body weights representative of children (age 6 mo-3 yrs), assumed to have an average body weight of 12 kg (normal distribution, std dev = 2.2).

Table 12-7: 95% Subjective Confidence Interval on the Meals of Fish from EFPC Consumed per Year that Would Have Yielded a Methylmercury Dose Equal to the RfD(based on *in utero* and adult exposure) [a, b]

Year	Adult exposure (compared to *in utero* RfD)			Adult exposure (compared to adult RfD)		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	2.5	5.0	13	7.5	15	38
1951	2.5	5.3	12	7.6	16	35
1952	2.6	5.0	12	7.7	15	35
1953	2.6	5.3	13	7.8	16	40
1954	2.6	5.3	12	7.8	16	36
1955	2.6	5.2	12	7.9	16	37
1956	2.6	5.5	12	7.7	16	37
1957	2.6	5.2	12	7.8	16	35
1958	2.5	5.2	12	7.4	16	36
1959	2.5	5.1	12	7.5	15	36
1960	2.5	5.4	12	7.6	16	37
1961	2.5	5.4	13	7.5	16	38
1962	2.5	5.1	12	7.5	15	36
1963	2.5	5.2	12	7.6	16	37
1964	2.7	5.2	11	8.0	16	34
1965	2.8	5.4	12	8.4	16	37
1966	2.7	5.5	13	8.0	16	39
1967	2.7	5.4	11	8.1	16	34
1968	2.9	5.3	11	8.7	16	34
1969	2.8	5.7	12	8.4	17	35
1970	2.8	5.5	13	8.4	17	38
1971	2.9	5.7	13	8.8	17	40
1972	2.9	5.9	12	8.7	18	37
1973	3.0	5.5	13	9.1	16	38
1974	3.1	5.6	14	9.2	17	41
1975	3.4	6.5	13	10	19	40
1976	3.3	6.3	15	10	19	45
1977	3.6	6.9	15	11	21	46
1978	3.7	7	17	11	21	50
1979	3.8	7.4	16	11	22	49
1980	3.7	7.5	17	11	22	50
1981	4.0	8.3	19	12	25	57
1982	4.0	7.9	18	12	24	54
1983	4.8	9	20	15	27	59
1984	5.6	10	21	17	30	63
1985	7.7	13	28	23	40	83
1986	8.0	14	31	34	41	94
1987	7.9	14	30	24	41	91
1988	7.8	13	27	23	39	82
1989	7.5	13	27	23	40	82
1990	7.6	14	30	23	41	89

a For the "*in utero* exposure" scenario, fish meal consumption rates equal to the RfD were calculated using the USEPA methylmercury RfD for *in utero* exposure of 0.0001 mg kg^{-1} d^{-1}. For the "adult exposure" scenario, fish meal consumption rates equal to the RfD were calculated using the adult exposure RfD of 0.0003 mg kg^{-1} d^{-1}.

b The 95%ile confidence intervals were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution, std dev = 9.5). 50^{th} %ile fish meal consumption rates for adult males equal to the RfD would likely be about 25% higher than the values presented under "adult exposure", due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

As shown in Table 12-6, if a small child consumed 1 meal of fish per year from EFPC between 1950 and 1975, there is a small probability that the child was exposed to methylmercury at an annual average dose equal to the USEPA RfD. If the child consumed 2 or more meals of fish per year from EFPC, it is likely that the child received a methylmercury dose in excess of the RfD. This RfD, however, is based on observations of neurological effects in children exposed *in utero*; it is not known whether very young children exposed to the same dose of methylmercury following birth are as sensitive to adverse health effects as if they were exposed *in utero*.

As shown in Table 12-7, for 1950-1975, if a pregnant adult female consumed fish from EFPC at a rate of approximately 3 or more fish meals per year, there is a small probability that she was exposed to methylmercury at a dose in excess of the USEPA RfD, and her unborn child may have been at risk of postnatal neurological effects from *in utero* exposure. If a pregnant adult female consumed fish from EFPC at a rate of 5 or more fish meals per year during these years, it is *likely* that she received a methylmercury dose in excess of the USEPA RfD. Table 12-7 also shows the 95% subjective confidence interval on the number of meals of fish from EFPC that an adult who is *not* pregnant would have had to consume per year to receive an annual average dose equal to the methylmercury RfD protective of adverse health effects in adults. As shown, a non-pregnant adult could consume a greater number of fish meals per year without being at risk of adverse health effects from methylmercury exposure.

Population Size

As discussed in Section 6.2, the estimated total size of the Scarboro Community population during a given year was assumed to be 800 to 1,200 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 400 to 600 individuals), adult females of child-bearing age comprised approximately 20% of the population (or about 160 to 240 individuals), and children ages 6 months to 3 years comprised approximately 15% of the population (or about 120 to 180 individuals).

Conclusions

Based on this evaluation, mercury doses to residents of the Scarboro Community may have exceeded the USEPA RfDs if individuals spent a significant amount of time in or near EFPC during the years of highest mercury releases from Y-12 (1953-1962), consumed homegrown fruits and vegetables during the years 1952-1965, or consumed one or more meals of EFPC fish per year (children) or three or more meals of EFPC fish per year (adults). In addition, children who lived in Scarboro during the mid- to late-1950s may have been exposed to elemental mercury through inhalation at doses that exceeded the elemental mercury RfD. Given the relatively large size of the potentially exposed population and the relatively high doses estimated for the fruit/vegetable consumption, EFPC water and sediment contact, and fish consumption pathways, it is likely that some individuals who lived in the Scarboro Community between 1953 and 1962 were exposed to mercury at doses that exceeded the USEPA RfDs.

12.1.3 Robertsville School Students, 1950-1990

The Robertsville School Students population reflects exposures to children ages 12 to 15 years who attended Robertsville Junior High School. These students were assumed to be exposed to mercury through inhalation and through dermal contact and incidental ingestion of schoolyard soil. In addition to exposures during school hours, a small number of students were assumed to occasionally recreate in EFPC and have been exposed to mercury through dermal contact and ingestion of EFPC sediment and water. Examples of the types of activities these children may have engaged in include wading, swimming, and fishing. Figures 12-11 and 12-12 show estimated elemental mercury and total inorganic mercury doses for general students for 1950-1990. Figures 12-13 and 12-14 show estimated elemental mercury and total inorganic mercury doses for student-recreators. The highest doses were estimated to have been in 1957 and 1958, since these were the years that mercury releases to EFPC from Y-12 were estimated to have been greatest.

Table 12-8 summarizes the years that the annual average elemental and inorganic mercury doses exceeded the USEPA RfD or the NOAELs, and presents the ratios of the highest estimated doses to the RfDs and NOAELs. Ratios in excess of 1.0 indicate that the RfD or NOAEL was exceeded.

Exposures to Elemental Mercury

The estimated elemental mercury doses (from inhalation) were below the USEPA RfD for all years evaluated.

Exposures to Inorganic mercury

The estimated total inorganic mercury doses (from all of the remaining pathways combined) exceeded the USEPA inorganic mercury RfD for the following years:

- *General student**S** Upper bounds* on the estimated total inorganic mercury doses exceeded the RfD for 1955-1956 and 1958. Doses for these years were less than 1/300 of the NOAEL.

- *Recreational users of EFPC**S** Upper bounds* on the estimated total inorganic mercury doses exceeded the RfD for 1955-1958. Doses for these years were less than 1/200 of the NOAEL.

Inorganic mercury doses slightly above the NOAEL have been associated with kidney effects in laboratory rodents. Health effects in humans exposed to inorganic mercury at doses at or below the NOAEL have not been reported.

Figure 12-11: Robertsville School Children (General Student)-
Comparison of Estimated Elemental Mercury Doses
from INHALATION of Airborne Mercury
with Toxicity Benchmark Values for Elemental Mercury

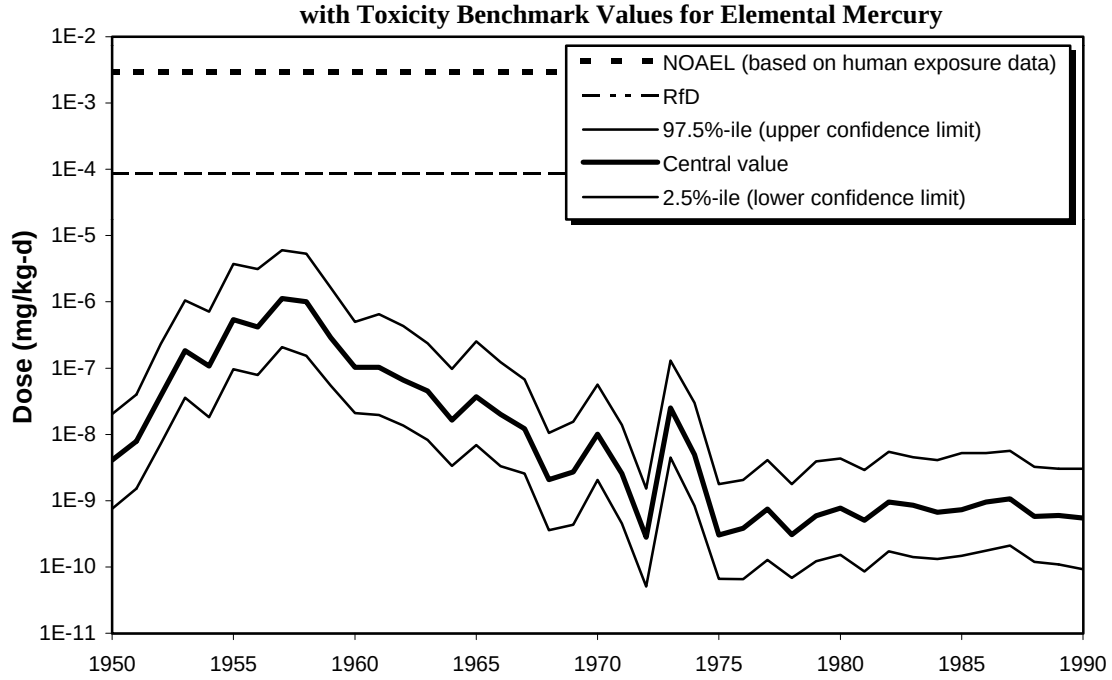


Figure 12-12: Robertsville School Children (General Students)-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses
with Toxicity Benchmark Values for Inorganic Mercury

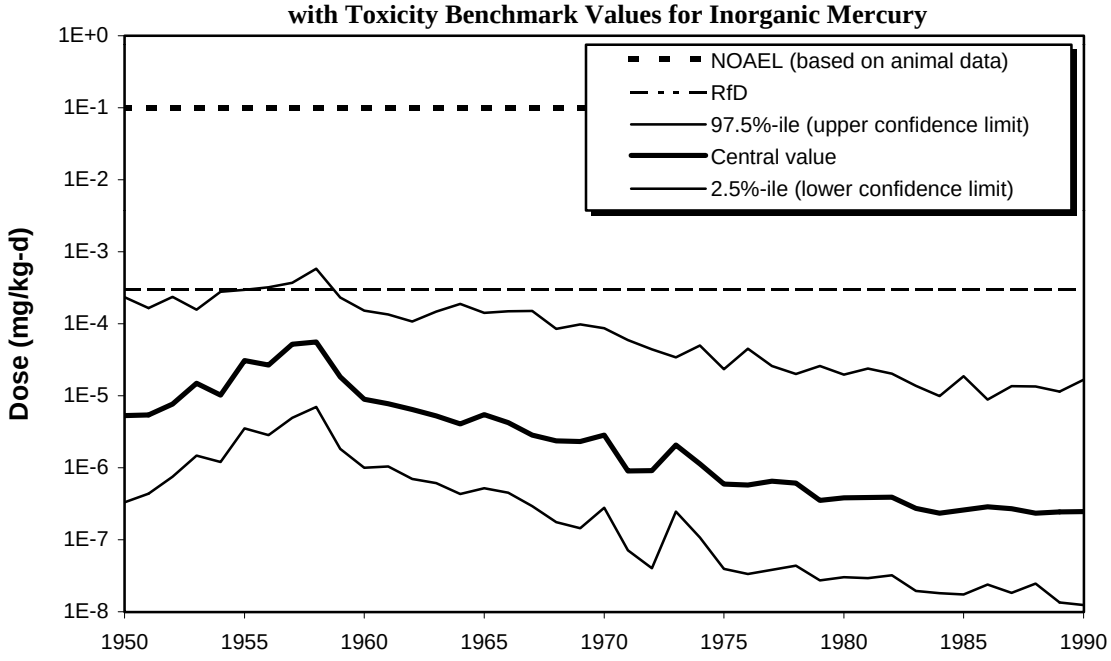


Figure 12-13: Robertsville School Children (EFPC Recreator)-
Comparison of Estimated Elemental Mercury Doses
from INHALATION of Airborne Mercury
with Toxicity Benchmark Values for Elemental Mercury

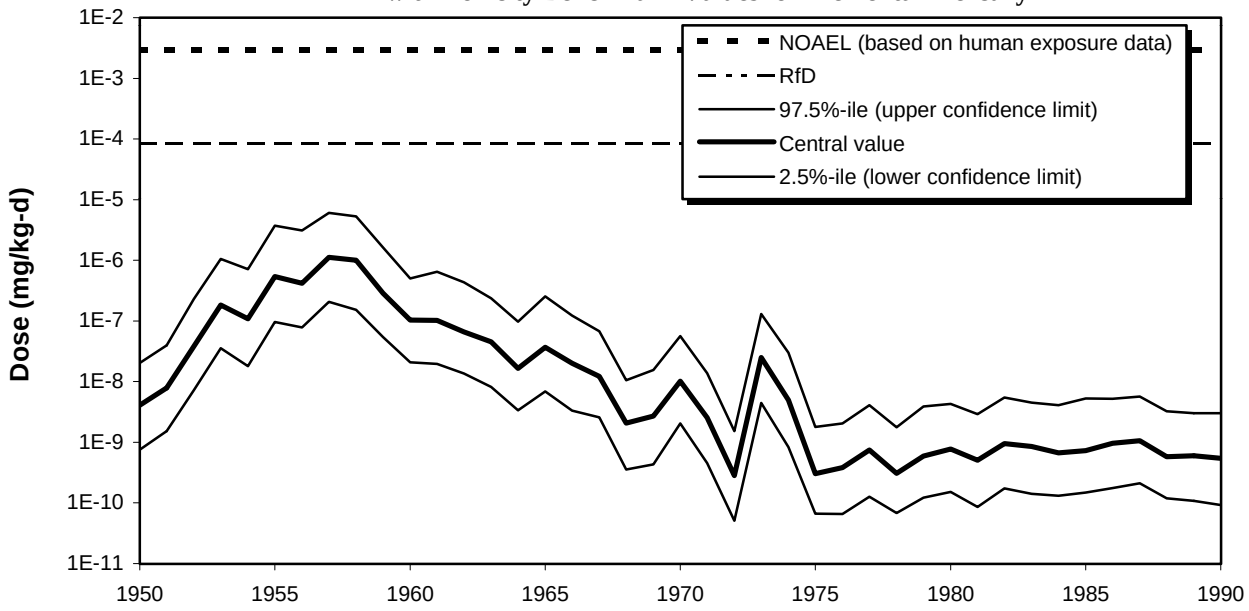


Figure 12-14: Robertsville School Children (EFPC Recreator)-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses
with Toxicity Benchmark Values for Inorganic Mercury

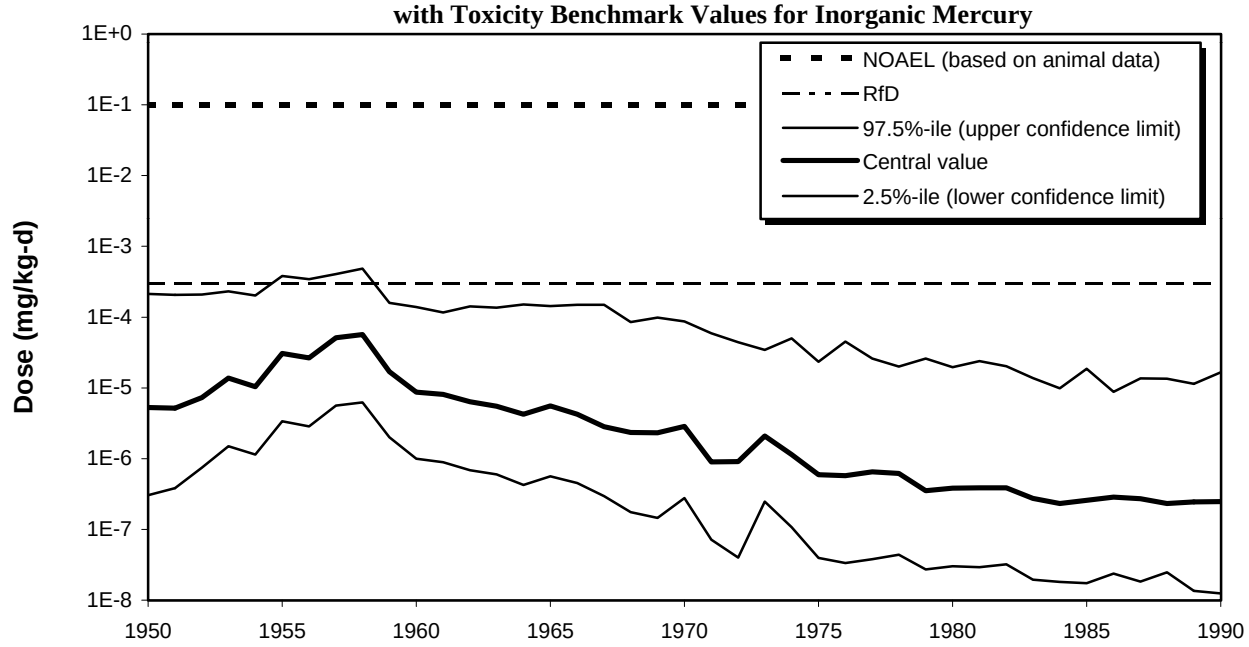


Table 12-8: Robertsville School StudentsS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	Estimated Population Size [a]	%-ile	Years Dose Exceeded the:		Ratio of Highest Dose (at given %-ile) to:	
			USEPA RfD [b]	NOAEL [c]	USEPA RfD [b]	NOAEL [c]
General Student						
Elemental mercury	1500-2000	97.5%ile	---	---	0.070 *(1957)*	0.0021 *(1957)*
		50%ile	---	---	0.013 *(1957)*	0.00038 *(1957)*
		2.5%ile	---	---	0.0011 *(1955)*	0.000033 *(1955)*
Inorganic mercury	1500-2000	97.5%ile	1955-56, 1958	---	**1.1** *(1958)*	0.0033 *(1958)*
		50%ile	---	---	0.013 *(1957)*	0.000040 *(1957)*
		2.5%ile	---	---	0.00017 *(1958)*	0.00000051 *(1958)*
Student-EFPC Recreator						
Elemental mercury	5-20	97.5%ile	---	---	0.070 *(1957)*	0.041 *(1957)*
		50%ile	---	---	0.013 *(1957)*	0.0076 *(1957)*
		2.5%ile	---	---	0.0011 *(1955)*	0.0021 *(1955)*
Inorganic mercury	5-20	97.5%ile	1955-58	---	**1.6** *(1958)*	0.0048 *(1958)*
		50%ile	---	---	0.19 *(1958)*	0.00057 *(1958)*
		2.5%ile	---	---	0.021 *(1958)*	0.000062 *(1958)*

a Annual average population size
b USEPA RfD for elemental mercury = 0.000086 mg kg^{-1} d^{-1}; USEPA RfD for inorganic mercury = 0.00030 mg kg^{-1} d^{-1}
c NOAEL for elemental mercury = 0.0029 mg kg^{-1} d^{-1}; NOAEL for inorganic mercury = 0.10 mg kg^{-1} d^{-1}

As shown in Figure 10-4, the exposure pathway with the greatest contribution to the estimated total inorganic mercury dose for general students was skin contact with contaminated floodplain soil. In 1958, 96% of the mean total inorganic mercury dose was estimated to be contributed by this pathway. For the EFPC recreator, the dominant contributors to the estimated total inorganic mercury dose were skin contact with EFPC water and skin contact with soil (contributing approximately 42% and 36%, respectively, of the estimated mean total inorganic mercury dose for 1958).

Sensitivity analyses show that the dominant contributor to variance in estimated doses from contact with contaminated soil is the soil concentration. The rate of soil loading on skin also contributes significantly to variance. Because historical measurements of mercury concentrations in floodplain soil near Robertsville School are not available, historical soil concentrations were estimated based on concentrations measured

in homogenized 0-16 inch core samples collected in 1991-1992 and factors applied to the "averaged" homogenized concentration to account for the likelihood that surface soil concentrations were higher during the period of highest mercury releases from Y-12. Soil concentration estimates were thus associated with a significant degree of uncertainty.

For the skin contact with EFPC water pathway, the dominant contributors to variance are the exposure time and exposure frequency to surface water and the skin permeability rate.

Population Size

As discussed in Section 6.2, the estimated total size of the Robertsville School general student population (boys and girls ages 12 to 15 years) during a given year was assumed to be 1,500 to 2,000 students. The size of the recreational-users of EFPC population during a given year was assumed to be 5 to 20 students.

Conclusions

Based on this evaluation, mercury doses to Robertsville School students may have exceeded the USEPA RfDs if individuals came in contact with contaminated surface soil in the school yard near the creek, or spent a significant amount of time in or near EFPC during the years of highest mercury releases from Y-12 (1953-1962). Because of the relatively large size of this population, it is likely that some individuals in this population were exposed to mercury at doses that exceeded the RfD.

12.1.4 EFPC Floodplain Farm Family, 1950-1990

The EFPC Floodplain Farm Family population reflects exposures to individuals who lived in rural farms along the EFPC floodplain. These individuals were assumed to be exposed to mercury through inhalation; dermal contact with soil and EFPC sediment and water; and ingestion of soil, EFPC sediment and water, "backyard" meat, milk, and fruits/vegetables, and EFPC fish. Figures 12-15 and 12-16 show estimated elemental mercury doses (from inhalation) for adults and children for 1950-1990. Figures 12-17 and 12-18 show total estimated inorganic mercury doses for all of the inorganic mercury pathways combined (the remaining pathways with the exclusion of fish consumption). Figures 12-19 and 12-20 show estimated methylmercury doses from consumption of EFPC fish. The highest doses were estimated to have been in 1957 and 1958, since these were the years that mercury releases to EFPC from Y-12 were estimated to be greatest.

Table 12-9 summarizes the years that the annual average elemental, inorganic, and methyl mercury doses exceeded the USEPA RfD or NOAEL, and presents the ratios of the highest estimated doses to the RfDs and NOAELs. Ratios in excess of 1.0 indicate that the RfD or NOAEL was exceeded.

Figure 12-15: EFPC Floodplain Farm Family-
Comparison of Estimated Elemental Mercury Doses (Adult)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

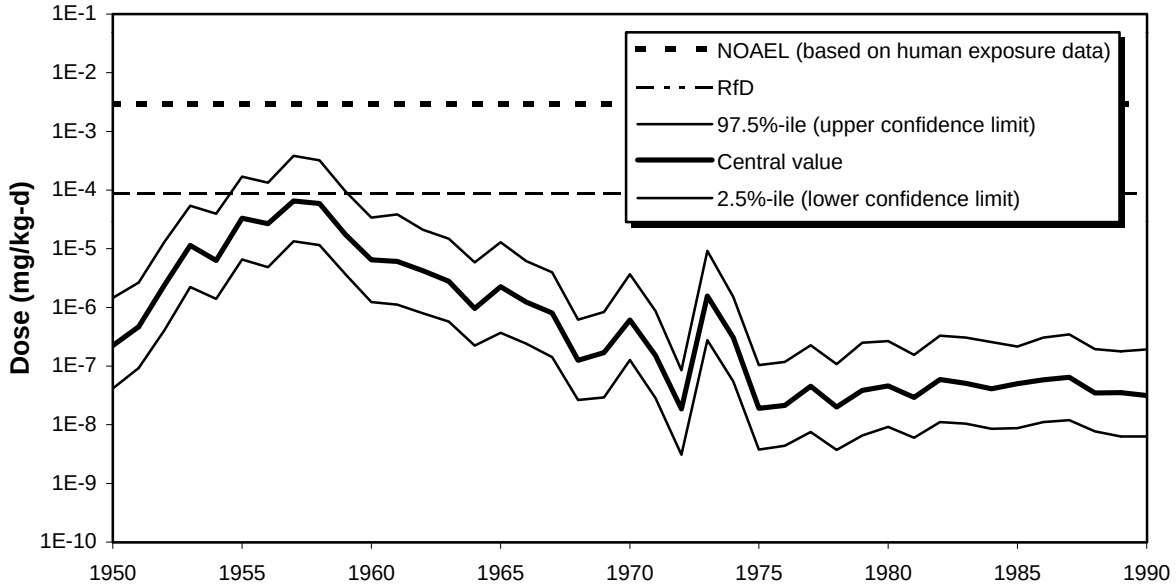


Figure 12-16: EFPC Floodplain Farm Family-
Comparison of Estimated Elemental Mercury Doses (Child)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

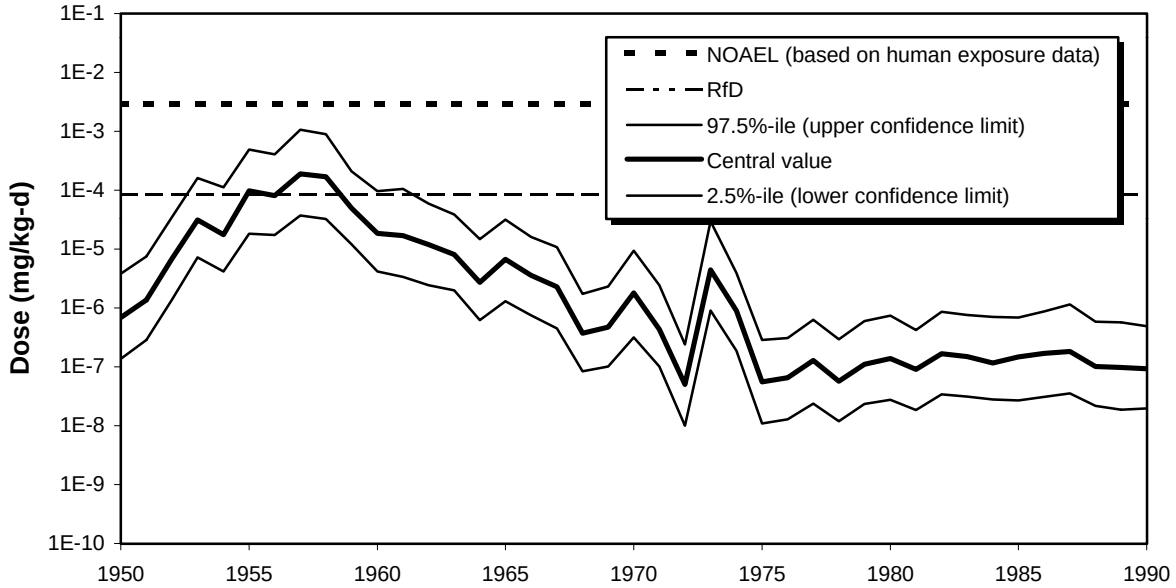


Figure 12-17: EFPC Floodplain Farm Family-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Adult)
with Toxicity Benchmark Values for Inorganic Mercury

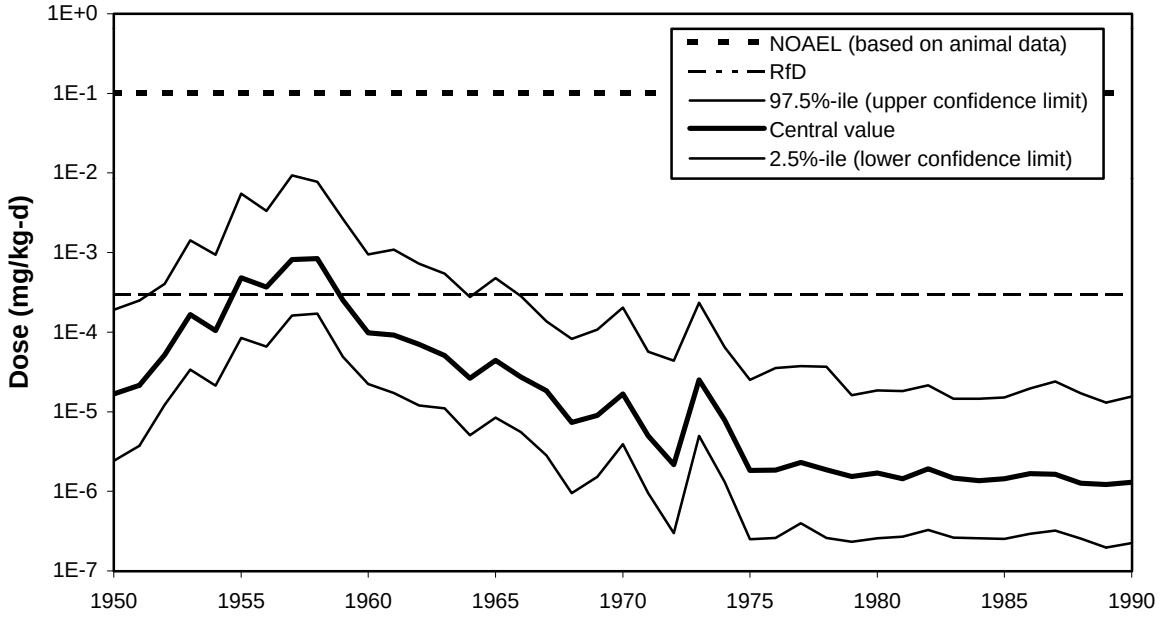


Figure 12-18: EFPC Floodplain Farm Family-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Child)
with Toxicity Benchmark Values for Inorganic Mercury

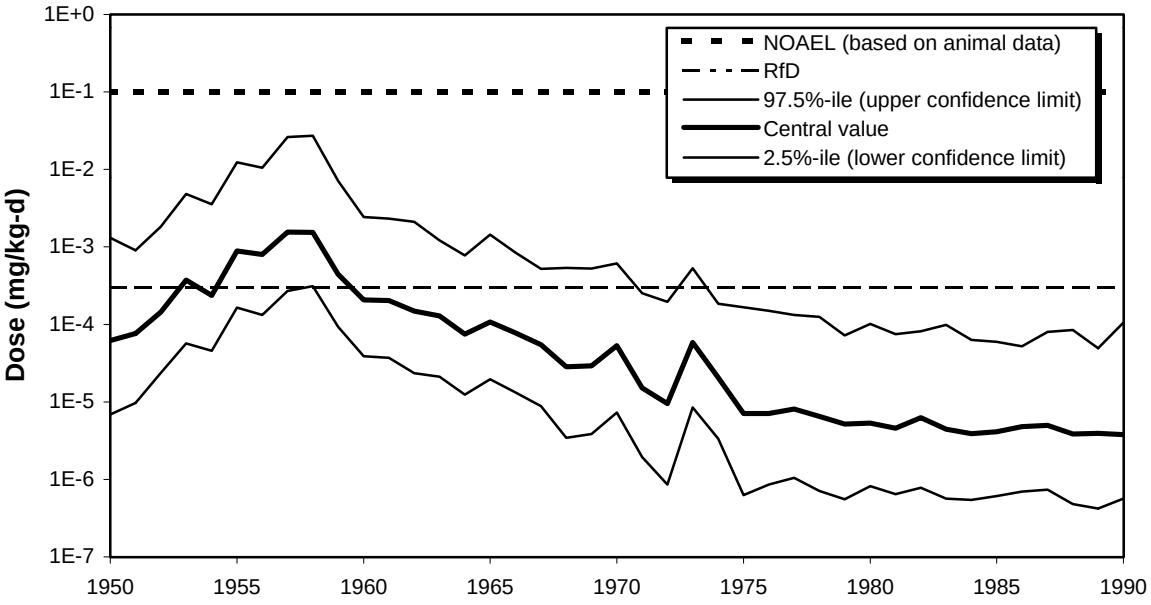


Figure 12-19: EFPC Floodplain Farm Family-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

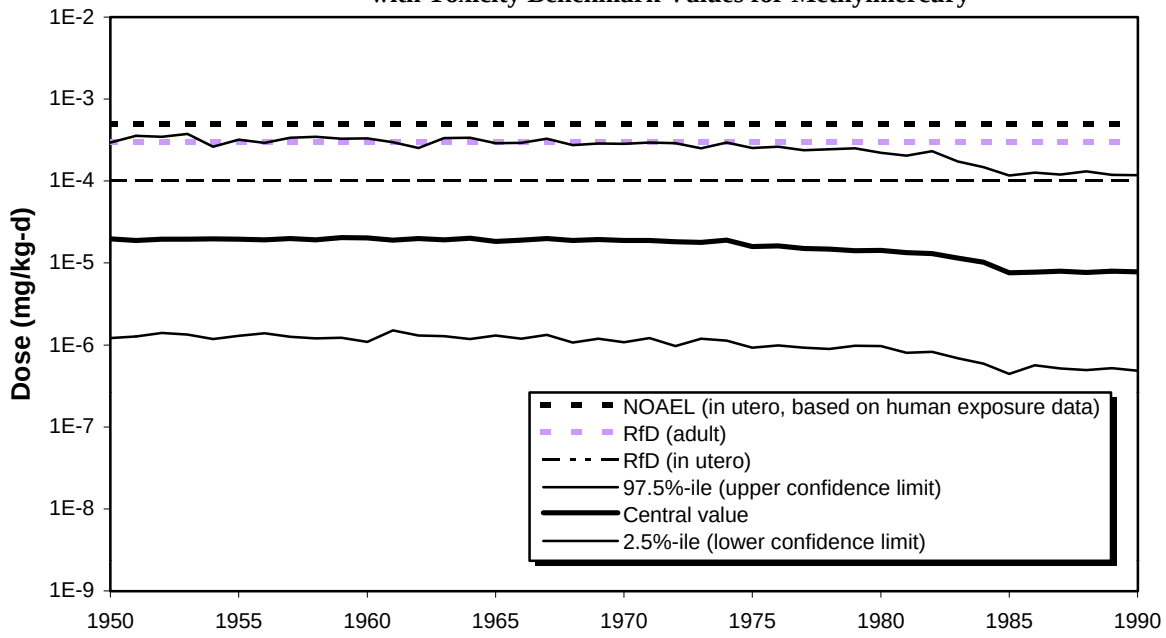


Figure 12-20: EFPC Floodplain Farm Family-
Comparison of Estimated Methylmercury Doses (Child)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

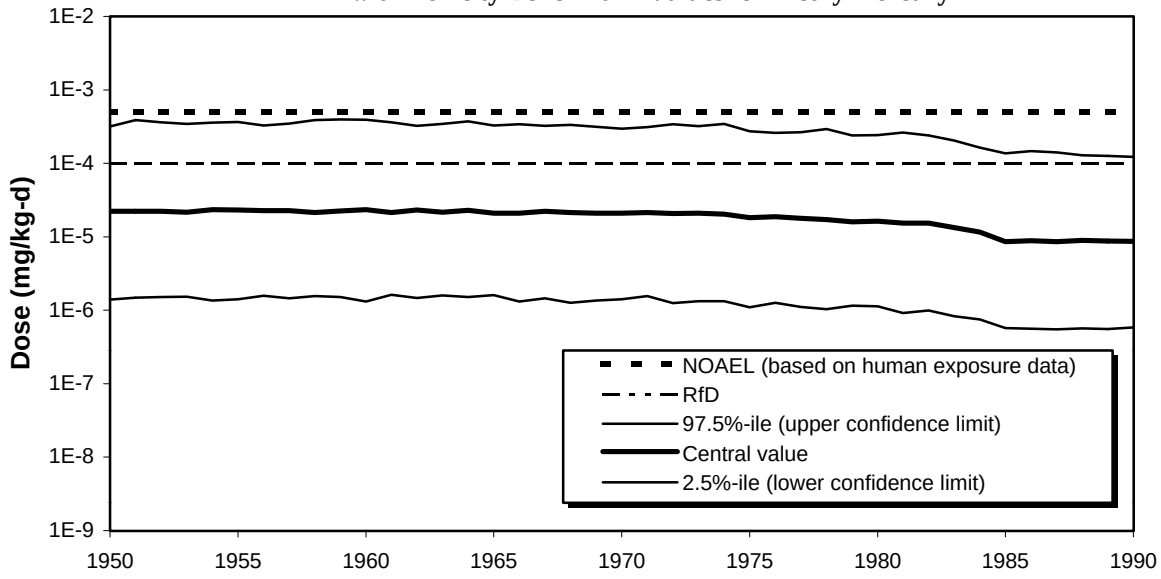


Table 12-9: EFPC Floodplain Farm FamilyS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	Estimated Population Size [a]	%-ile	Years Dose Exceeded the:		Ratio of Highest Dose (at given %-ile) to:	
			USEPA RfD [b]	NOAEL [c]	USEPA RfD [b]	NOAEL [c]
Adult						
Elemental mercury	5-10	97.5%ile	1955-59	---	**4.4** *(1957)*	0.013 *(1957)*
		50%ile	---	---	0.76 *(1957)*	0.022 *(1957)*
		2.5%ile	---	---	0.15 *(1957)*	0.0045 *(1957)*
Inorganic mercury	5-10	97.5%ile	1952-65	---	**31** *(1957)*	0.094 *(1957)*
		50%ile	1955-58	---	**2.8** *(1958)*	0.0084 *(1958)*
		2.5%ile	---	---	0.57 *(1958)*	0.0017 *(1958)*
Methyl-mercury	2-4, 5-10	97.5%ile	1950-90 [1954-73] [d]	---	**3.6 [1.2]** [d] *(1957)*	0.72 *(1957)*
		50%ile	---	---	0.21 [0.07] [d] *(1955)*	0.042 *(1955)*
		2.5%ile	---	---	0.013 [0.0043] [d] *(1957)*	0.0026 *(1957)*
Child (6 mo - 3 yrs)						
Elemental mercury	2-8	97.5%ile	1953-61	---	**13** *(1957)*	0.38 *(1957)*
		50%ile	1955, 1957-58	---	2.2 *(1957)*	0.066 *(1957)*
		2.5%ile	---	---	0.43 *(1957)*	0.013 *(1957)*
Inorganic mercury	2-8	97.5%ile	1950-70, 1973	---	**90** *(1958)*	0.27 *(1958)*
		50%ile	1953, 1955-59	---	**5.0** *(1958)*	0.015 *(1958)*
		2.5%ile	1958	---	**1.0** *(1958)*	0.0031 *(1958)*
Methyl-mercury	2-8	97.5%ile	1950-90	---	3.9 *(1956)*	0.78 *(1956)*
		50%ile	---	---	0.24 *(1956)*	0.048 *(1956)*
		2.5%ile	---	---	0.014 *(1956)*	0.0028 *(1956)*

a Annual average population size; for methylmercury adult exposure, the first range is for women of child-bearing age and the second range is for all adults

b USEPA RfD for elemental mercury = 0.000086 mg kg^{-1} d^{-1}; USEPA RfD for inorganic mercury = 0.00030 mg kg^{-1} d^{-1}; USEPA RfD for methylmercury (*in utero* exposure)= 0.00010 mg kg^{-1} d^{-1}; RfD for methylmercury (adult exposure) = 0.00030 mg kg^{-1} d^{-1}

c NOAEL for elemental mercury = 0.0029 mg kg^{-1} d^{-1}; NOAEL for inorganic mercury = 0.10 mg kg^{-1} d^{-1}; NOAEL for methylmercury (*in utero* exposure) = 0.0005 mg kg^{-1} d^{-1}

d First value represents comparison to *in utero* exposure RfD, second value represents comparison to adult exposure RfD

Exposures to Elemental Mercury

The estimated elemental mercury doses (from inhalation) exceeded the USEPA elemental mercury RfD for the following years:

- *Adults***S** *Upper bounds* on the estimated total elemental mercury doses exceeded the RfD for 1955-1959. Doses for these years were less than 1/7 of the NOAEL.

- *Children***S** *Upper bounds* on the estimated elemental mercury doses exceeded the RfD for 1953-1961; *central values* exceeded the RfD for 1955 and 1957-1958. Doses for these years ranged from about 1/3 to 1/30 of the NOAEL.

Elemental mercury doses slightly above the NOAEL have been associated with neurological effects in adult workers exposed to airborne mercury**S** the neurological effects observed included hand tremor, increases in memory disturbances, and evidence of dysfunction of the autonomic (involuntary) nervous system (IRIS 1998). At slightly higher doses, evidence of effects on the kidney have also been observed. Health effects in humans exposed to elemental mercury at doses at or below the NOAEL have not been reported.

Sensitivity analyses showed that the dominant contributor to variance in the inhalation dose estimates was the air concentration term. Airborne mercury concentrations at the location of the EFPC Floodplain Farm Family were assumed to result from volatilization of mercury from EFPC surface water to air. Air concentration estimates were associated with a significant degree of uncertainty because of the lack of any direct measurements of airborne mercury and uncertainties about how much mercury could have volatilized from EFPC. These uncertainties could not be resolved within the constraints of this project; consequently, wide uncertainty bounds were placed on the parameters describing the atmospheric transport of mercury.

Exposures to Inorganic mercury

The estimated total inorganic mercury doses (from all inorganic mercury pathways combined) exceeded the USEPA inorganic mercury RfD for the following years:

- *Adults***S** *Upper bounds* on the estimated total inorganic mercury doses exceeded the RfD for 1952-1963 and 1965; *central values* exceeded the RfD for 1955-1958. *Upper bounds* on estimated doses for these years were less than 1/10 of the NOAEL.

- *Children***S** *Upper bounds* on the estimated total inorganic mercury doses exceeded the RfD for 1950-1970 and 1973; *central values* exceeded the RfD for 1953 and 1955-1959. The *lower bound* on the estimated total inorganic mercury dose for 1958 was at the RfD, and below the RfD for all other years. Doses for these years were less than 1/3 of the NOAEL.

Inorganic mercury doses slightly above the NOAEL have been associated with kidney effects in laboratory rodents. Health effects in humans exposed to inorganic mercury at doses at or below the NOAEL have not been reported.

As shown in Figure 10-4, the exposure pathway with the greatest contribution to the estimated total inorganic mercury dose for the EFPC Floodplain Farm Family residents was ingestion of homegrown above-ground fruits and vegetables contaminated by airborne mercury. In 1957, 79% of the mean total inorganic mercury dose to adults was estimated to be contributed by this pathway; for children, the contribution was approximately 88%. Other pathways contributing significantly to the child total inorganic mercury dose were incidental ingestion of EFPC surface water, dermal contact with EFPC surface water, and ingestion of vegetables contaminated by soil mercury (contributing 11%, 3%, and 2%, respectively, during 1957 and yielding estimated inorganic mercury doses at the 97.5th percentile during 1955-1958 that were at or slightly above the USEPA inorganic mercury RfD).

Sensitivity analyses show that the dominant contributor to variance in estimated doses from ingestion of fruits/vegetables contaminated by airborne mercury is the ingestion rate. Parameters also contributing significantly to variance are the air concentration and factors describing deposition of airborne mercury to plant surfaces (detailed results of the sensitivity analysis are summarized in Appendix Y). For the contact with EFPC surface water pathways, the dominant contributors to variance are the surface water ingestion rate, duration and frequency of exposure, and the surface area of exposed skin.

Table 12-10 shows the 95% subjective confidence interval on the amount of homegrown fruits/vegetables that a child or adult would have had to consume in a given year to be exposed to inorganic mercury at an annual average dose equal to the USEPA RfD. As shown, during the years of highest mercury releases from Y-12 (1953 to 1962), children or adults consuming even moderate amounts of homegrown fruits or vegetables may have been exposed to mercury at doses exceeding the RfD. For most other years, it is unlikely that EFPC Floodplain Farm Family members would have been exposed to mercury through the homegrown fruit/vegetable pathway at doses exceeding the USEPA RfD, because it is unlikely that an individual would have consumed sufficiently large amounts of homegrown fruits and vegetables.

Table 12-10: 95% Subjective Confidence Intervals on the Pounds of Homegrown Above-Ground Fruits/Vegetables (fresh weight) Consumed per Year that Would Have Yielded an Inorganic Mercury Dose Equal to the USEPA RfDS EFPC Floodplain Farm Family [a]

Year	Child (6 mo - 3 yrs)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	90	630	>2,000	300	>2,000	>2,000
1951	32	290	>2,000	170	1,300	>2,000
1952	6.2	54	610	36	340	>2,000
1953	1.6	14	140	8.9	67	720
1954	3.1	22	220	11	110	1,100
1955	0.54	3.9	51	3.0	22	210
1956	0.59	5.5	48	3.2	28	250
1957	0.21	1.9	19	1.1	10	100
1958	0.29	2.4	22	1.7	13	150
1959	0.94	8.5	100	5.0	40	450
1960	2.4	22	210	15	130	1,100
1961	2.6	22	260	18	130	1,300
1962	4.3	33	390	20	160	1,500
1963	5.5	53	760	33	280	>2,000
1964	18	140	1,100	91	750	>2,000
1965	8.9	63	510	43	360	>2,000
1966	16	110	1,000	90	610	>2,000
1967	24	180	1,800	120	980	>2,000
1968	120	1,000	>2,000	720	>2,000	>2,000
1969	99	830	>2,000	540	>2,000	>2,000
1970	31	240	>2,000	160	1,200	>2,000
1971	110	950	>2,000	580	>2,000	>2,000
1972	1,200	>2,000	>2,000	>2,000	>2,000	>2,000
1973	9.4	89	1,100	57	550	>2,000
1974	60	450	>2,000	360	>2,000	>2,000
1975	720	>2,000	>2,000	>2,000	>2,000	>2,000
1976	810	>2,000	>2,000	>2,000	>2,000	>2,000
1977	410	>2,000	>2,000	1,800	>2,000	>2,000
1978	770	>2,000	>2,000	>2,000	>2,000	>2,000
1979	560	>2,000	>2,000	>2,000	>2,000	>2,000
1980	330	>2,000	>2,000	1,900	>2,000	>2,000
1981	500	>2,000	>2,000	>2,000	>2,000	>2,000
1982	310	>2,000	>2,000	1,800	>2,000	>2,000
1983	320	>2,000	>2,000	2,000	>2,000	>2,000
1984	420	>2,000	>2,000	>2,000	>2,000	>2,000
1985	360	>2,000	>2,000	1,700	>2,000	>2,000

Year	Child (6 mo - 3 yrs)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1986	330	>2,000	>2,000	1,700	>2,000	>2,000
1987	250	>2,000	>2,000	1,300	>2,000	>2,000
1988	520	>2,000	>2,000	>2,000	>2,000	>2,000
1989	400	>2,000	>2,000	>2,000	>2,000	>2,000
1990	670	>2,000	>2,000	>2,000	>2,000	>2,000

a Estimated consumption rates greater than 2,000 pounds per year (i.e., about 5.5 pounds per day, fresh weight) are considered highly unlikely, and are indicated as "> 2,000".

b The 95%ile confidence intervals for adults were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution). 50th %ile fruit/vegetable consumption rates for adult males equal to the RfD would likely be about 25% higher due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

Table 12-11 shows the 95% subjective confidence interval on the number of hours per year that a child or adult would have had to have been exposed to EFPC surface water (assuming that exposure occurs through both incidental ingestion and dermal contact) to have received an annual average inorganic mercury dose equal to the USEPA RfD. As shown, for most years, a child would have had to spend a very large number of hours in the creek to have been exposed to an inorganic mercury dose exceeding the RfD. During the years of highest mercury releases to EFPC (1957 and 1958), a child who spent about 15 or more hours per year in the creek may have been exposed to mercury at doses exceeding the RfD. During 1953-1956 and 1959-1963, children who played in the creek frequently (more than 30 hours per year) may have been exposed to mercury at doses exceeding of the RfD.

Tables 12-10 and 12-11 present 95% subjective confidence intervals of ingestion rates or exposure times that would have resulted in doses that exceeded the RfD if these were the *only* pathways through which the individual was exposed. If an individual consumed homegrown fruits/vegetables *and* played in EFPC (which for this population is highly likely), both pathways would contribute to the total inorganic mercury dose, and lower fruit/vegetable ingestion rates or exposure times to EFPC water than shown in Table 12-10 and 12-11 could have resulted in total inorganic mercury doses from both sources combined that exceeded the RfD.

Table 12-11: 95% Subjective Confidence Intervals on the Hours of Exposure to EFPC Surface Water per Year that Would have Yielded an Inorganic Mercury Dose Equal to the USEPA RfDS EFPC Floodplain Farm Family [a]

Year	Child (6 mo - 3 yr)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	4,300	>8,760	>8,760	>8,760	>8,760	>8,760
1951	2,000	5,900	>8,760	8,400	>8,760	>8,760
1952	360	1,000	4,600	1,500	4,000	>8,760
1953	75	240	990	320	1,000	5,100
1954	150	430	2,000	560	1,700	7,000
1955	29	79	430	110	320	1,600
1956	38	110	500	150	440	1,800
1957	18	39	150	60	160	650
1958	15	32	150	60	140	490
1959	56	130	500	210	490	1,700
1960	170	360	1,400	540	1,400	5,000
1961	190	420	1,600	770	1,800	6,400
1962	290	740	3,200	1,100	3,000	>8,760
1963	330	890	3,900	1,300	3,600	>8,760
1964	790	2,000	8,700	3,100	8,200	>8,760
1965	360	910	4,000	1,300	3,700	>8,760
1966	680	2,000	7,400	2,700	7,700	>8,760
1967	1,300	3,400	>8,760	5,200	>8,760	>8,760
1968	6,900	>8,760	>8,760	>8,760	>8,760	>8,760
1969	6,200	>8,760	>8,760	>8,760	>8,760	>8,760
1970	1,500	3,300	>8,760	6,000	>8,760	>8,760
1971	6,700	>8,760	>8,760	>8,760	>8,760	>8,760
1972	>8,760	>8,760	>8,760	>8,760	>8,760	>8,760
1973	730	1,600	6,900	3,000	7,000	>8,760
1974	3,400	7,100	>8,760	>8,760	>8,760	>8,760
>1974 (per year)	>8,760	>8,760	>8,760	>8,760	>8,760	>8,760

a Estimated exposure times that exceed the number of hours in a year are indicated as ">8,760"

b The 95%ile confidence intervals for adults were calculated using body weights and skin surface areas representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution, std dev = 9.5) and an average skin surface area of 8000 cm^2 (lognormal distribution, std dev = 800). 50^{th} %ile hours for adult males equal to the RfD would likely be about 10-15% higher due to their higher average body weight and skin surface area (the average body weight of an adult male is about 78 kg and the average skin surface area is about 10,000 cm^2, USEPA 1995).

Exposures to Methylmercury

The *upper bound* (97.5th percentile) of the estimated methylmercury doses for adults and children from consumption of fish caught in EFPC exceeded the USEPA methylmercury RfD (based on *in utero* exposures) for all years. However, for all years, adult and child doses were less than the NOAEL. Methylmercury doses slightly above the NOAEL have been associated with observations of neurological effects in children who were exposed to methylmercury *in utero* (their mothers consumed methylmercury in fish).

Table 12-6, discussed in Section 12.1.2, shows the 95% subjective confidence interval on the number of meals of fish from EFPC that a child would have had to consume per year to receive an annual average methylmercury dose equal to the USEPA RfD for methylmercury (based on *in utero* exposures, assumed to be protective of health effects in developing young children). Table 12-7 shows the 95% subjective confidence interval on the number of meals of fish from EFPC an adult female would have had to consume per year to receive an annual average methylmercury doses equal to the RfDs for methylmercury based on *in utero* and adult exposures.

As shown in Table 12-6, if a small child consumed 1 meal of fish per year from EFPC between 1950 and 1975, there is a small probability that the child was exposed to methylmercury at an annual average dose equal to the USEPA RfD. If the child consumed 2 or more meals of fish per year from EFPC, it is likely that the child received a methylmercury dose in excess of the RfD. As shown in Table 12-7, for 1950-1975, if a pregnant adult female consumed fish from EFPC at a rate of approximately 2 to 3 fish meals per year, there is a small probability that she was exposed to methylmercury at a dose in excess of the USEPA RfD, and her unborn child may have been at risk of postnatal neurological effects from *in utero* exposure. If a pregnant adult female consumed fish from EFPC at a rate of 5 or more fish meals per year during these years, it is *likely* that she was exposed to methylmercury at a dose in excess of the USEPA RfD.

Population Size

As discussed in Section 6.2, the estimated total size of the EFPC Floodplain Farm Family population during a given year was assumed to be 10 to 50 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 5 to 25 individuals), adult females of child-bearing age comprised approximately 20% of the population (or about 2 to 10 individuals) and children ages 6 months to 3 years comprised approximately 15% of the population (or about 1 to 8 individuals).

Conclusions

Based on this evaluation, mercury doses to members of the EFPC Floodplain Farm Family may have exceeded the USEPA RfDs if individuals spent a significant amount of time in or near EFPC during the years of highest mercury releases from Y-12 (1953-1962), consumed large amounts of homegrown fruits and vegetables during the years 1952-1965, or consumed one or more meals of EFPC fish per year

(children) or three or more meals of EFPC fish per year (adults). In addition, adults and children who lived along EFPC during the 1950s and early 1960s may have been exposed to elemental mercury through inhalation at doses that exceeded the elemental mercury RfD. Although the size of the potentially exposed population was small, because of the close proximity of this population to EFPC and the floodplain, the relatively high doses estimated for the fruit/vegetable consumption, EFPC water and sediment contact, and fish consumption pathways, and the likelihood that these families had backyard gardens and came in frequent contact with EFPC, it is likely that some individuals who lived on farms along the EFPC floodplain between 1950 and 1975 were exposed to mercury at doses that exceeded the USEPA RfDs.

12.1.5 Community Populations, 1950-1990

The Oak Ridge Community populations reflect exposures to individuals who lived in west Oak Ridge further from EFPC than the "farm families" (who were assumed to live on the edge of the floodplain), but within a mile of EFPC. These individuals were assumed to be exposed to mercury through inhalation and ingestion of fruits and vegetables from backyard gardens. As discussed in Section 6.2.1.5, the location of Community Population #1 was assumed to be approximately 200 yards north of the floodplain, near the intersection of Louisiana Avenue and Lincoln Road. The location of Community Population #2 was assumed to be approximately one-quarter mile from EFPC off Jefferson Avenue between Robertsville Road and Livingston Road. While specific locations were selected for purposes of air modeling, exposures estimated for Community Populations #1 and #2 were assumed to be representative of people who lived in residential areas within ½-mile and 1-mile of the creek, respectively.

Figures 12-21 and 12-22 show estimated elemental mercury doses (from inhalation) for Community Population #1 adults and children for 1950-1990; Figures 12-23 and 12-24 show total estimated inorganic mercury doses for this population. Figures 12-25 and 12-26 show estimated elemental mercury doses for Community Population #2 adults and children for 1950-1990; Figures 12-27 and 12-28 show total estimated inorganic mercury doses for this population. The highest doses were estimated to have been in 1957, since this was the year that estimated mercury releases to EFPC from Y-12 were greatest.

Table 12-12 summarizes the years that the estimated annual average elemental and inorganic mercury doses exceeded the USEPA RfD or the NOAELs, and presents the ratios of the highest estimated doses to the RfDs and NOAELs. Ratios in excess of 1.0 indicate that the RfD or NOAEL was exceeded.

Figure 12-21: Oak Ridge Community Resident #1-
Comparison of Estimated Elemental Mercury Doses (Adult)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

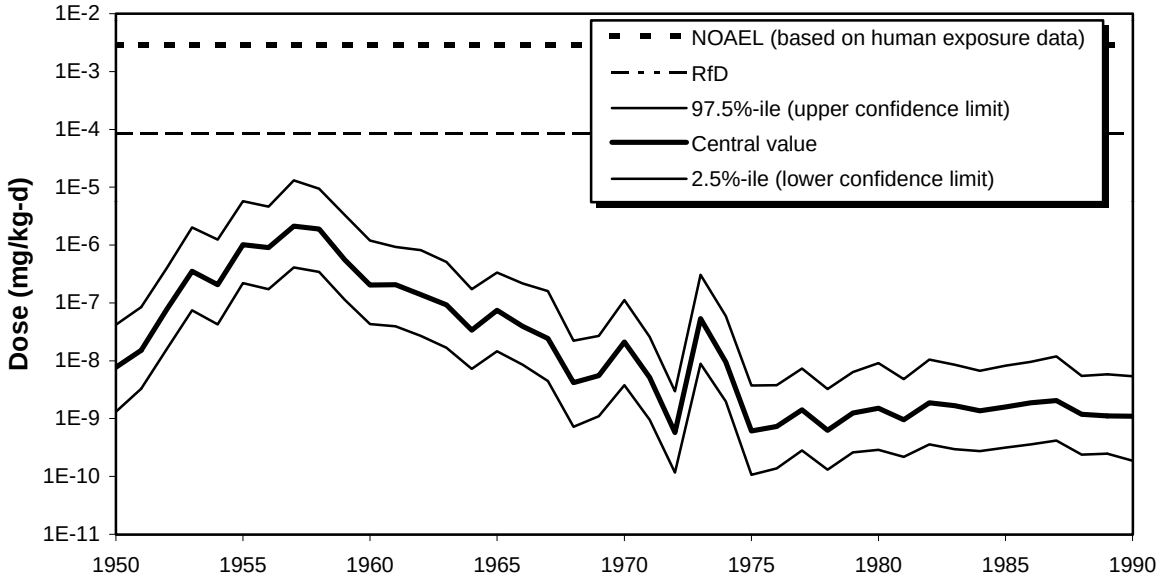


Figure 12-22: Oak Ridge Community Resident #1-
Comparison of Estimated Elemental Mercury Doses (Child)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

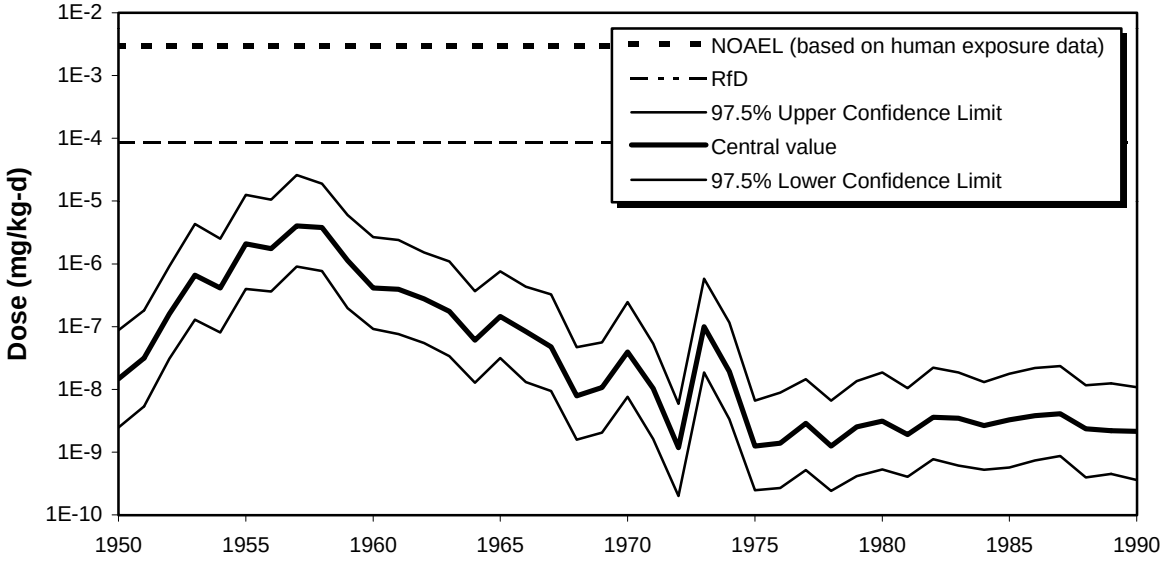


Figure 12-23: Oak Ridge Community Resident #1-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Adult)
with Toxicity Benchmark Values for Inorganic Mercury

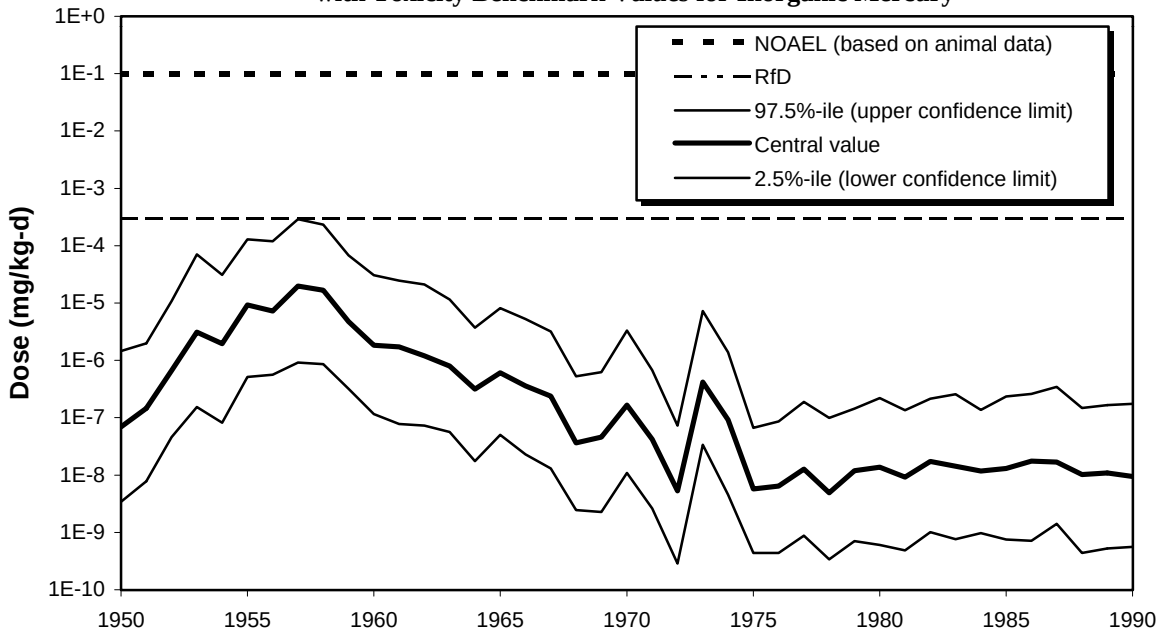


Figure 12-24: Oak Ridge Community Resident #1-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Child)
with Toxicity Benchmark Values for Inorganic Mercury

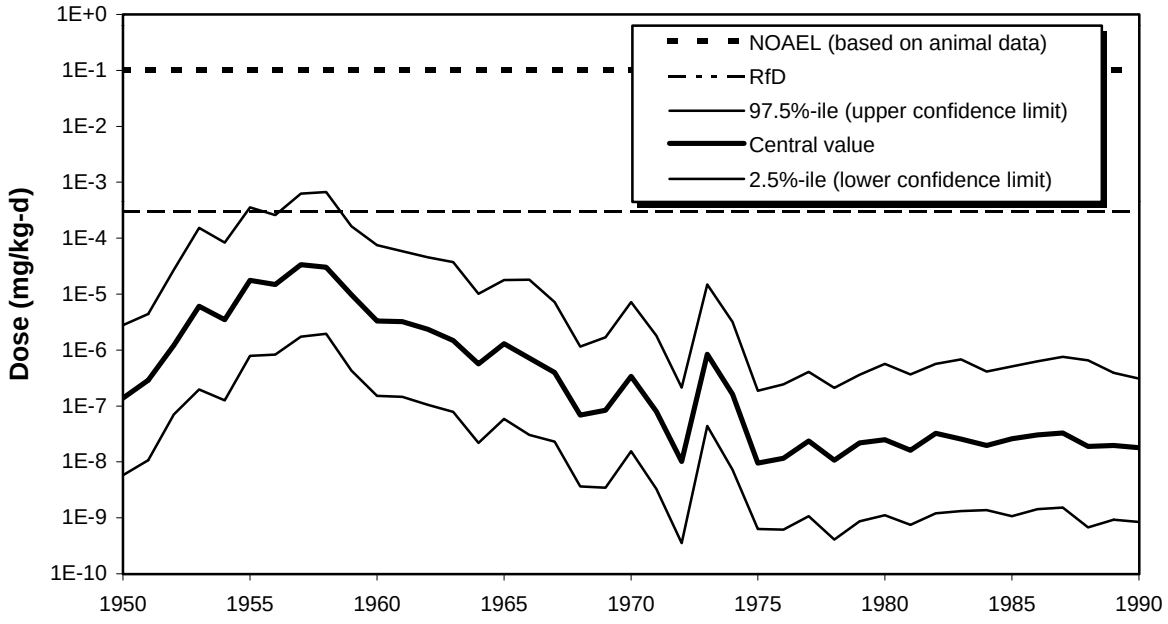


Figure 12-25: Oak Ridge Community Resident #2-
Comparison of Estimated Elemental Mercury Doses (Adult)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

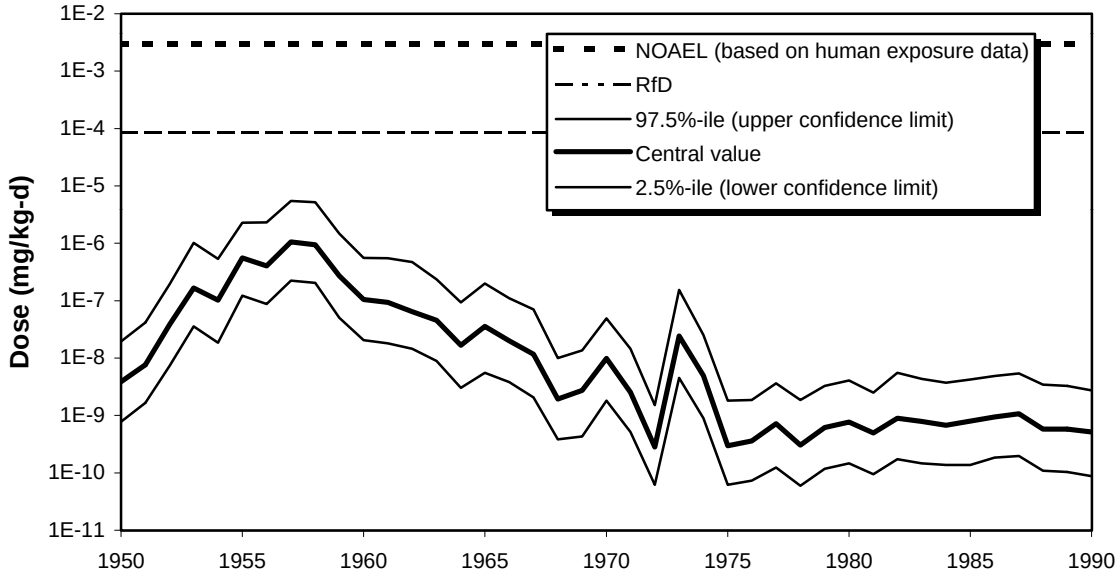


Figure 12-26: Oak Ridge Community Resident #2-
Comparison of Estimated Elemental Mercury Doses (Child)
from INHALATION OF AIRBORNE MERCURY
with Toxicity Benchmark Values for Elemental Mercury

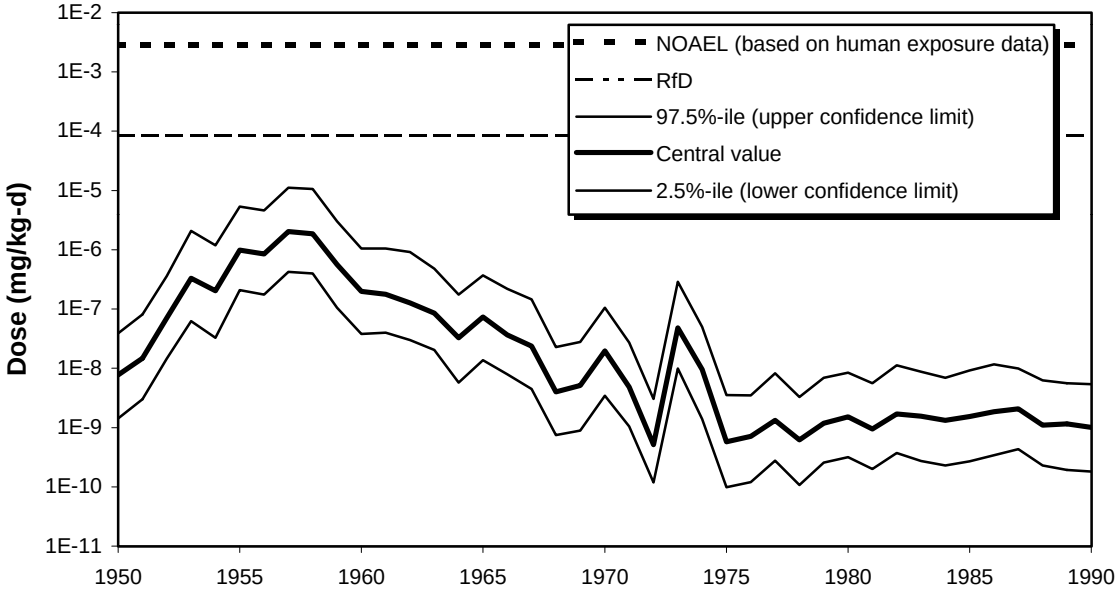


Figure 12-27: Oak Ridge Community Resident #2-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Adult)
with Toxicity Benchmark Values for Inorganic Mercury

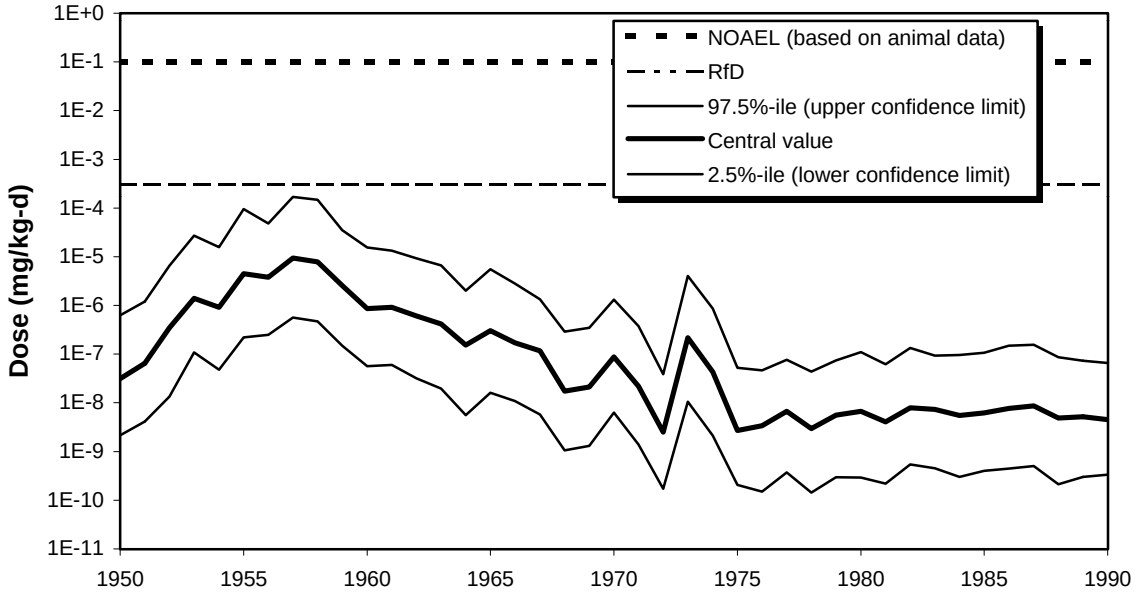


Figure 12-28: Oak Ridge Community Resident #2-
Comparison of Estimated TOTAL INORGANIC MERCURY Doses (Child)
with Toxicity Benchmark Values for Inorganic Mercury

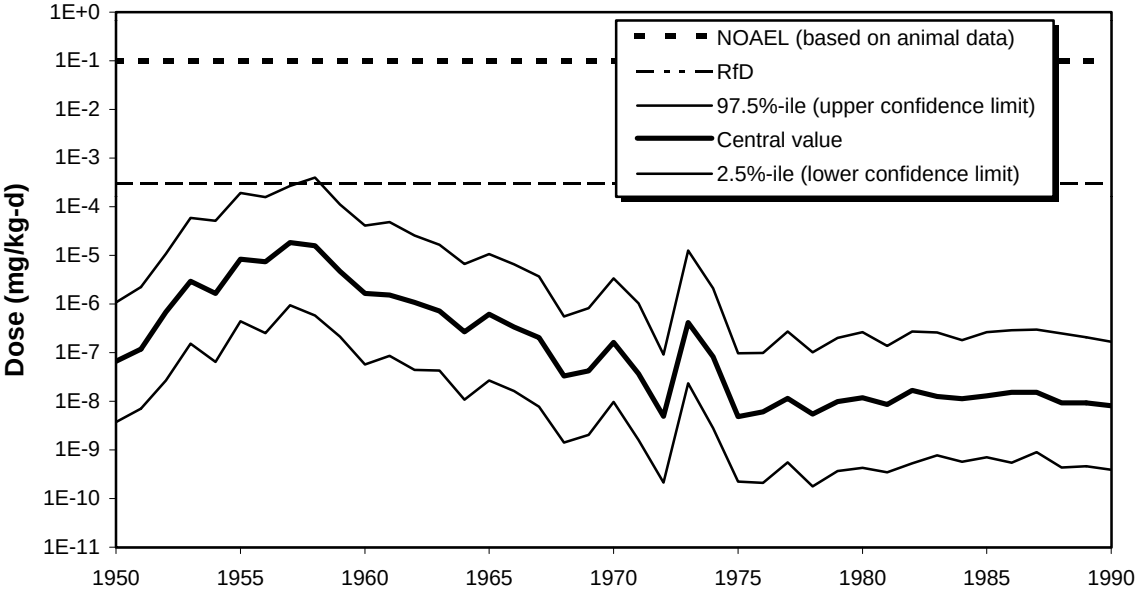


Table 12-12: Community PopulationsS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	Estimated Population Size [a]	%-ile	Years Dose Exceeded the:		Ratio of Highest Dose (at given %-ile) to:	
			USEPA RfD [b]	NOAEL [c]	USEPA RfD [b]	NOAEL [c]
Community Population #1						
Adult						
Elemental mercury	1000-1500	97.5%ile	---	---	0.15 *(1957)*	0.0045 *(1957)*
		50%ile	---	---	0.024 *(1957)*	0.00072 *(1957)*
		2.5%ile	---	---	0.0048 *(1957)*	0.00014 *(1957)*
Inorganic mercury	1000-1500	97.5%ile	---	---	0.30 *(1957)*	0.0029 *(1957)*
		50%ile	---	---	0.067 *(1957)*	0.00020 *(1957)*
		2.5%ile	---	---	0.0031 *(1957)*	0.0000092 *(1957)*
Child (6 mo-3yrs)						
Elemental mercury	300-450	97.5%ile	---	---	0.30 *(1957)*	0.0090 *(1957)*
		50%ile	---	---	0.047 *(1957)*	0.0014 *(1957)*
		2.5%ile	---	---	0.011 *(1957)*	0.00031 *(1957)*
Inorganic mercury	300-450	97.5%ile	1955, 1957-58	---	**2.2** *(1957)*	0.0067 *(1957)*
		50%ile	---	---	0.11 *(1957)*	0.00034 *(1957)*
		2.5%ile	---	---	0.00063 *(1957)*	0.000019 *(1957)*
Community Population #2						
Adult						
Elemental mercury	1000-1500	97.5%ile	---	---	0.063 *(1957)*	0.0019 *(1957)*
		50%ile	---	---	0.013 *(1957)*	0.00038 *(1957)*
		2.5%ile	---	---	0.0026 *(1957)*	0.000076 *(1957)*
Inorganic mercury	1000-1500	97.5%ile	---	---	0.057 *(1957)*	0.0017 *(1957)*
		50%ile	---	---	0.031 *(1957)*	0.000094 *(1957)*
		2.5%ile	---	---	0.0019 *(1957)*	0.0000057 *(1957)*
Child (6 mo - 3 yrs)						
Elemental mercury	300-450	97.5%ile	---	---	0.13 *(1957)*	0.0038 *(1957)*
		50%ile	---	---	0.023 *(1957)*	0.00069 *(1957)*
		2.5%ile	---	---	0.0049 *(1957)*	0.00014 *(1957)*
Inorganic mercury	300-450	97.5%ile	1958	---	**1.3** *(1957)*	0.0040 *(1957)*
		50%ile	---	---	0.060 *(1957)*	0.00018 *(1957)*
		2.5%ile	---	---	0.0031 *(1957)*	0.0000094 *(1957)*

a Annual average population size
b USEPA RfD for elemental mercury = 0.000086 mg kg^{-1} d^{-1}; USEPA RfD for inorganic mercury = 0.00030 mg kg^{-1} d^{-1}
c NOAEL for elemental mercury = 0.0029 mg kg^{-1} d^{-1}; NOAEL for inorganic mercury = 0.10 mg kg^{-1} d^{-1}

The estimated mercury doses for adults and children of both populations were below the USEPA elemental mercury and inorganic mercury RfDs for all years, with the exception of exposures of Population #1 children to inorganic mercury in contaminated fruits and vegetables in 1955 and 1957-1958 and exposures of Population #2 children to inorganic mercury in fruits and vegetables for 1958.

Tables 12-13 and 12-14 show the 95% subjective confidence intervals on the amount of homegrown fruits/vegetables that Community Population #1 and Population #2 children or adults would have had to consume in a given year to be exposed to inorganic mercury at an annual average dose equal to the USEPA RfD. As shown, during the years of highest mercury releases from Y-12 (1953 to 1962), children or adults who lived close to the floodplain and consumed moderate amounts of homegrown above-ground fruits or vegetables (more than about 10 pounds per year for children and about 40 pounds per year for adults) may have been exposed to mercury at doses exceeding the USEPA RfD for inorganic mercury. If these individuals consumed 60 pounds per year (children) or 300 pounds per year (adults) of homegrown above-ground fruits or vegetables, it is likely that they were exposed to mercury at doses exceeding the RfD. Individuals living further from EFPC would have received lower mercury doses through this pathway per pound of fruits/vegetables consumed. For most other years, it is unlikely that Community residents would have been exposed to mercury through the homegrown fruit/vegetable pathway at doses exceeding the USEPA RfD, because it is unlikely that an individual would have consumed sufficiently large amounts of homegrown fruits and vegetables.

Population Size

As discussed in Section 6.2, the estimated total size of the Community Populations during a given year was assumed to be 2,000 to 3,000 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 1,000 to 1,500 individuals), adult females of child-bearing age comprised approximately 20% of the population (or about 400 to 600 individuals) and children ages 6 months to 3 years comprised approximately 15% of the population (or about 300 to 450 individuals).

Conclusions

Based on this evaluation, members of the Oak Ridge Community populations who lived close to the EFPC floodplain may have been exposed to mercury at doses exceeding the USEPA RfDs if they consumed moderate amounts of homegrown fruits and vegetables during the years of highest airborne mercury releases from Y-12 (1953 to 1962). However, doses estimated for this pathway were below the RfD at the 50^{th} percentile of the distributions for all years, and below the 97.5^{th} percentile of the distributions for all but three years. Therefore, the number of individuals in the Community Populations who may have been exposed to inorganic mercury at doses exceeding the RfD, if any, was likely small.

Table 12-13: 95% Subjective Confidence Interval on the Pounds of Homegrown Above-Ground Fruits/Vegetables (fresh weight) Consumed per Year that Would Have Yielded an Inorganic Mercury Dose Equal to the USEPA RfDS Community Population #1 [a]

Year	Child (6 mo - 3 yr)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1951	1,300	>2,000	>2,000	>2,000	>2,000	>2,000
1952	180	2,000	>2,000	1,100	>2,000	>2,000
1953	46	420	>2,000	280	2,000	>2,000
1954	88	640	>2,000	470	>2,000	>2,000
1955	16	120	1,300	83	620	>2,000
1956	19	160	1,500	96	740	>2,000
1957	7.5	61	600	42	300	>2,000
1958	8.9	73	600	46	430	>2,000
1959	30	240	>2,000	180	1,200	>2,000
1960	75	600	>2,000	420	>2,000	>2,000
1961	85	670	>2,000	390	>2,000	>2,000
1962	140	1,000	>2,000	670	>2,000	>2,000
1963	170	1,500	>2,000	1,100	>2,000	>2,000
1964	470	>2,000	>2,000	>2,000	>2,000	>2,000
1965	260	1,900	>2,000	1,200	>2,000	>2,000
1966	480	>2,000	>2,000	>2,000	>2,000	>2,000
1967	600	>2,000	>2,000	>2,000	>2,000	>2,000
1968	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1969	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1970	990	>2,000	>2,000	>2,000	>2,000	>2,000
1971	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1972	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1973	360	>2,000	>2,000	1,400	>2,000	>2,000
1974	1,500	>2,000	>2,000	>2,000	>2,000	>2,000
>1974 (per year)	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000

a Estimated consumption rates greater than 2,000 pounds per year (i.e., about 5.5 pounds per day, fresh weight) are considered highly unlikely, and are indicated as "> 2,000".

b The 95%ile confidence intervals for adults were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution). 50^{th} %ile fruit/vegetable consumption rates for adult males equal to the RfD would likely be about 25% higher due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

Table 12-14: 95% Subjective Confidence Interval on the Pounds of Homegrown Above-Ground Fruits/Vegetables (fresh weight) Consumed per Year that Would Have Yielded an Inorganic Mercury Dose Equal to the USEPA RfDS Community Population #2 [a]

Year	Child (6 mo - 3 yr)			Adult Female [b]		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1951	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1952	430	>2,000	>2,000	>2,000	>2,000	>2,000
1953	99	770	>2,000	470	>2,000	>2,000
1954	140	1,300	>2,000	840	>2,000	>2,000
1955	32	250	>2,000	190	1,300	>2,000
1956	42	300	>2,000	290	1,700	>2,000
1957	14	130	1,300	80	670	>2,000
1958	17	160	1,300	84	690	>2,000
1959	67	470	>2,000	320	>2,000	>2,000
1960	160	1,400	>2,000	820	>2,000	>2,000
1961	180	1,500	>2,000	1,000	>2,000	>2,000
1962	230	2,000	>2,000	1,500	>2,000	>2,000
1963	330	>2,000	>2,000	1,800	>2,000	>2,000
1964	1,200	>2,000	>2,000	>2,000	>2,000	>2,000
1965	380	>2,000	>2,000	>2,000	>2,000	>2,000
1966	720	>2,000	>2,000	>2,000	>2,000	>2,000
1967	1,500	>2,000	>2,000	>2,000	>2,000	>2,000
1968	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1969	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1970	1,700	>2,000	>2,000	>2,000	>2,000	>2,000
1971	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1972	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
1973	790	>2,000	>2,000	>2,000	>2,000	>2,000
1974	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000
>1974 (per year)	>2,000	>2,000	>2,000	>2,000	>2,000	>2,000

a Estimated consumption rates greater than 2,000 pounds per year (i.e., about 5.5 pounds per day, fresh weight) are considered highly unlikely, and are indicated as "> 2,000".

b The 95%ile confidence intervals for adults were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution). 50th %ile fruit/vegetable consumption rates for adult males equal to the RfD would likely be about 25% higher due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

12.1.6 Fish- Consuming Populations

Mercury doses to people who ate fish from waterways downstream of Y-12 (Watts Bar Reservoir, Clinch River/Poplar Creek, and EFPC) were evaluated using two different approaches:

(1) 95% subjective confidence intervals on doses to populations who historically caught and consumed fish from these systems were reconstructed based on information about historical fish concentrations and assumptions about fishing habits**S** results from this evaluation can be used to assess the likelihood of existence of populations that were exposed to mercury in fish at levels exceeding health effects benchmark values, and

(2) 95% subjective confidence intervals on doses corresponding to three different levels of fish consumption were estimated for each system**S** results from this evaluation can be used by individuals who recall consuming fish from these systems, to determine whether they may have been at risk from methylmercury exposure.

Results from these two approaches are summarized below.

Doses to Historical Populations

Mercury doses from consumption of fish were estimated for four populations who fished in Watts Bar Reservoir or Clinch River/Poplar Creek:

- *Watts Bar Reservoir Commercial Anglers***S** adult members of these families were assumed to eat an average of 24 g of fish per day (range 0.97 **S** 90 g d^{-1}, lognormal distribution), equivalent to an average of 52 meals per year (range 2.1 **S** 190) assuming 170 g per fish meal,
- *Watts Bar Reservoir Recreational Anglers***S** adult members of these families were assumed to eat an average of 30 g of fish per day (range 1.2 **S** 110 g d^{-1}, lognormal distribution), equivalent to an average of 65 meals per year (range 2.6 **S** 240) assuming 170 g per fish meal,
- *Clinch River/Poplar Creek Commercial Anglers***S** adult members of these families were assumed to eat an average of 2.2 g of fish per day (range 0.090 **S** 8.4 g d^{-1}, lognormal distribution), equivalent to an average of 4.7 meals per year (range 0.19 **S** 18) assuming 170 g per fish meal, and
- *Clinch River/Poplar Creek Recreational Anglers***S** adult members of these families were assumed to eat an average of 18 g of fish per day (range 0.76 **S** 65 g d^{-1}, lognormal distribution), equivalent to an average of 39 meals per year (range 1.6 **S** 140) assuming 170 g per fish meal.

"Recreational anglers" include those who caught fish for personal consumption, while "commercial anglers" include full-time anglers who used commercial fishing gear to catch large amounts of fish for commercial sale and consumed fish they did not market. Child members of these families were assumed to eat about 1/5 the amount of fish as adults ate. Figures 12-29 through 12-36 show estimated annual average methylmercury doses for to these populations.

Table 12-15 summarizes the years that the estimated doses to Watts Bar Reservoir and Clinch River/ Poplar Creek recreational and commercial anglers from consumption of fish exceeded the USEPA RfD or the NOAEL for methylmercury, and presents the ratios of the highest estimated doses to the RfDs and NOAELs. Figure 12-37 shows how the mercury doses for all angler populations compare to the RfDs based on *in utero* and adult exposures and to the NOAEL at the 97.5^{th} (upper), 50^{th} (central), and 2.5^{th} (lower) percentiles of the dose distributions. Years that exceeded the RfDs are indicated with light shading, and years that exceeded the NOAEL are indicated with dark shading.

In applying the above approach to estimate concentrations in fish based on sediment core measurements, results were not constrained to be at or near zero in the early 1950s.

In using the results of the dose reconstruction for fish consumption from Watts Bar Reservoir, Clinch River/Poplar Creek, and EFPC, one should keep in mind that results for the early 1950s have likely been overestimated to some degree because elevated concentrations were estimated beginning in 1950, while concentrations in fish likely built up in a more gradual manner over time. During the early 1950s, releases from the Y-12 Plant to EFPC had begun, but were quite low compared to releases that occurred after 1952. The elevated concentrations in these early years are most likely due to imprecision of the analysis and "dating" of concentrations in sediment cores that were used to estimate concentrations in fish.

Figure 12-29: Watts Bar Commercial Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

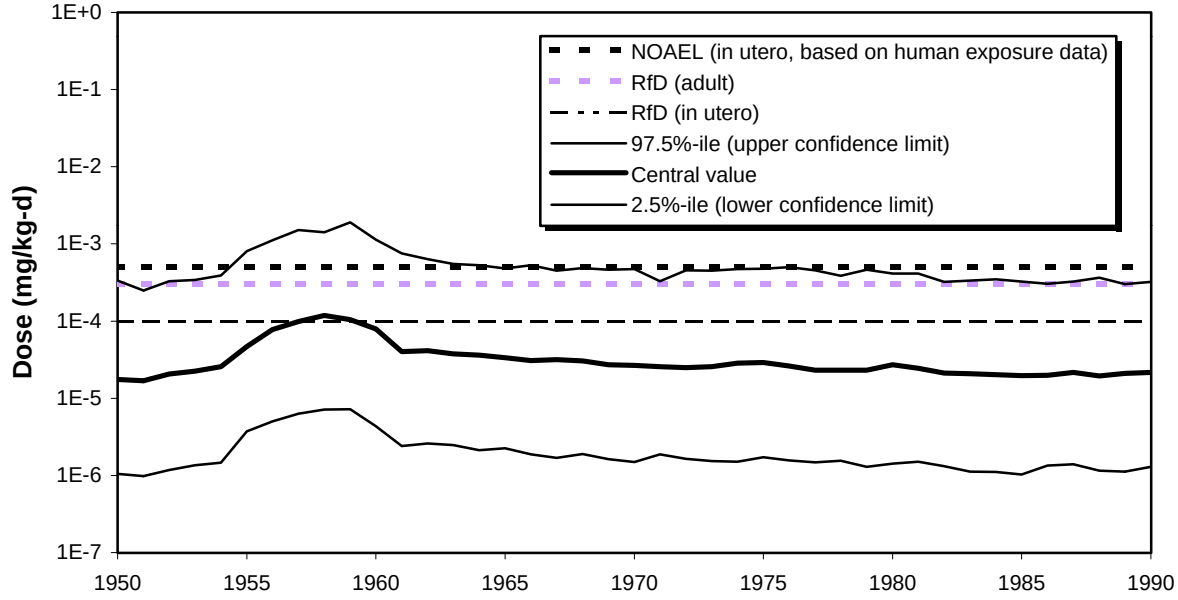


Figure 12-30: Watts Bar Commercial Anglers-
Comparison of Estimated Methylmercury Doses (Child)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

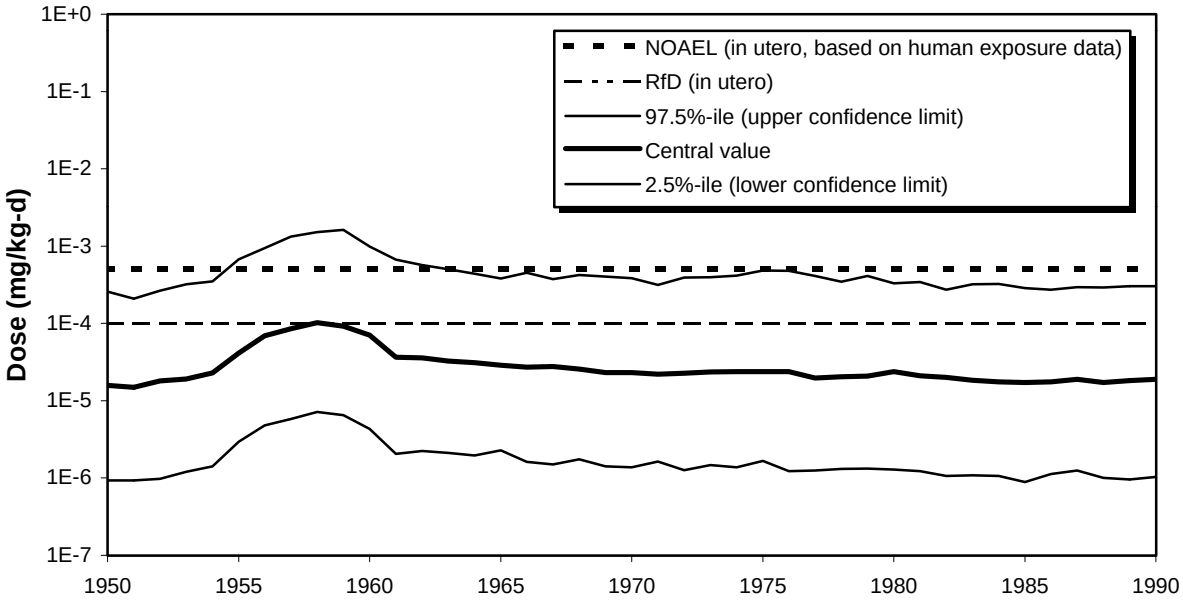


Figure 12-31: Watts Bar Recreational Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

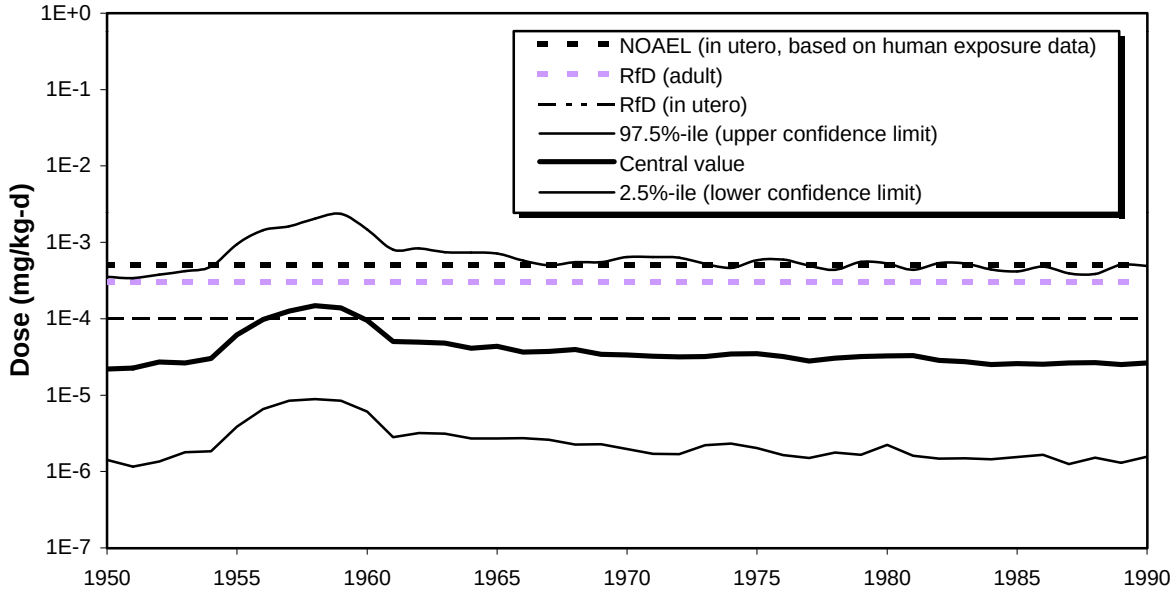


Figure 12-32: Watts Bar Recreational Anglers-
Comparison of Estimated Methylmercury Doses (Child)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

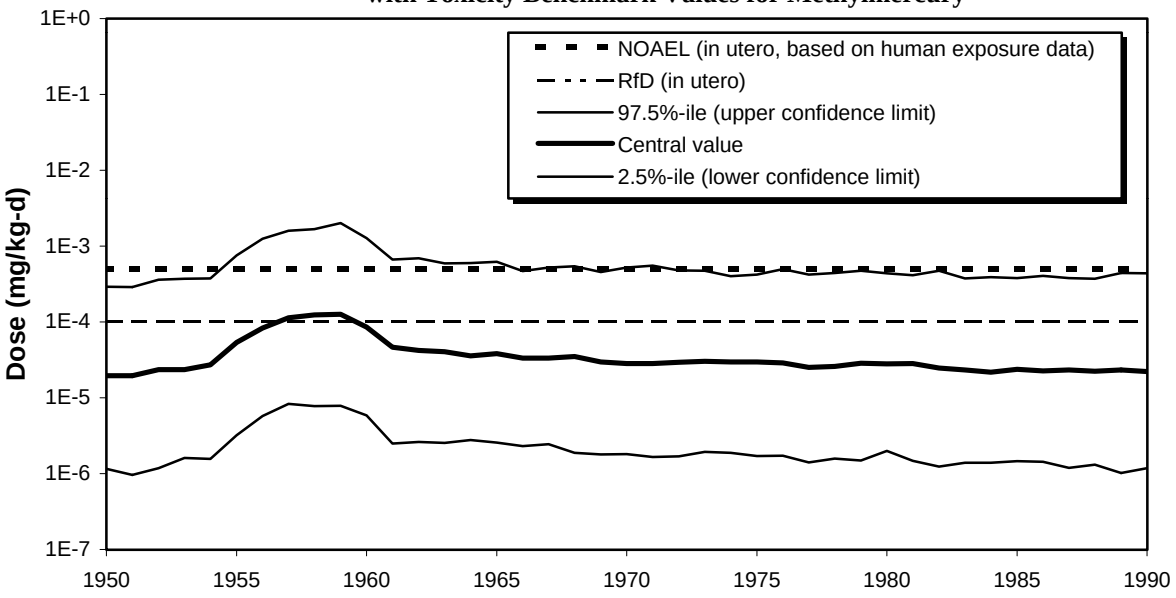


Figure 12-33: Clinch River/Poplar Creek Commercial Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

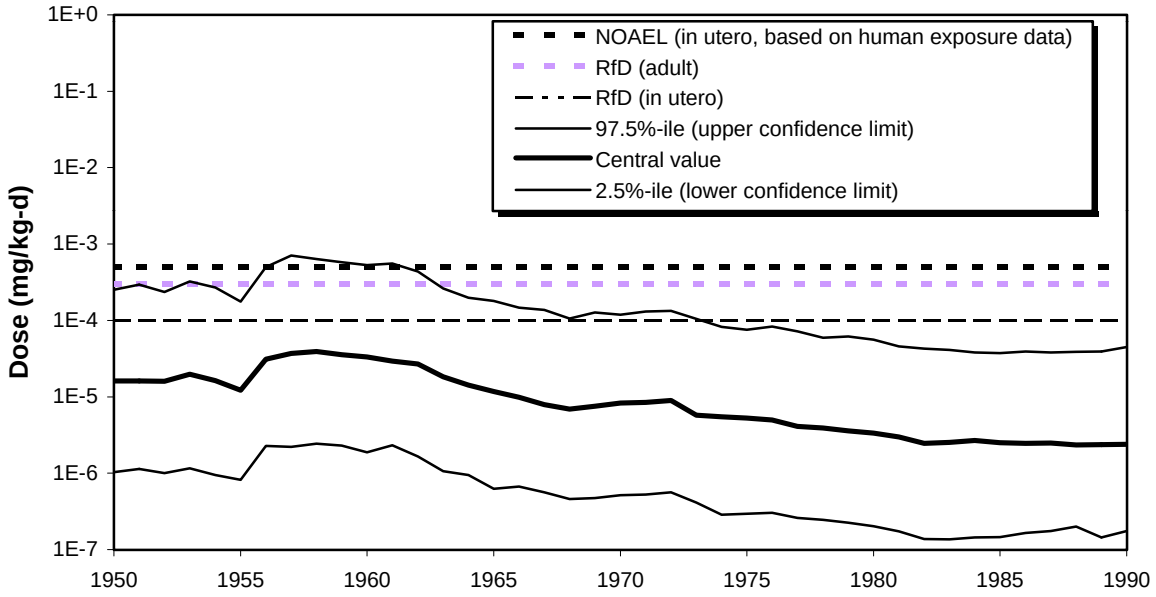


Figure 12-34: Clinch River/Poplar Creek Commercial Anglers-
Comparison of Estimated Methylmercury Doses (Child)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

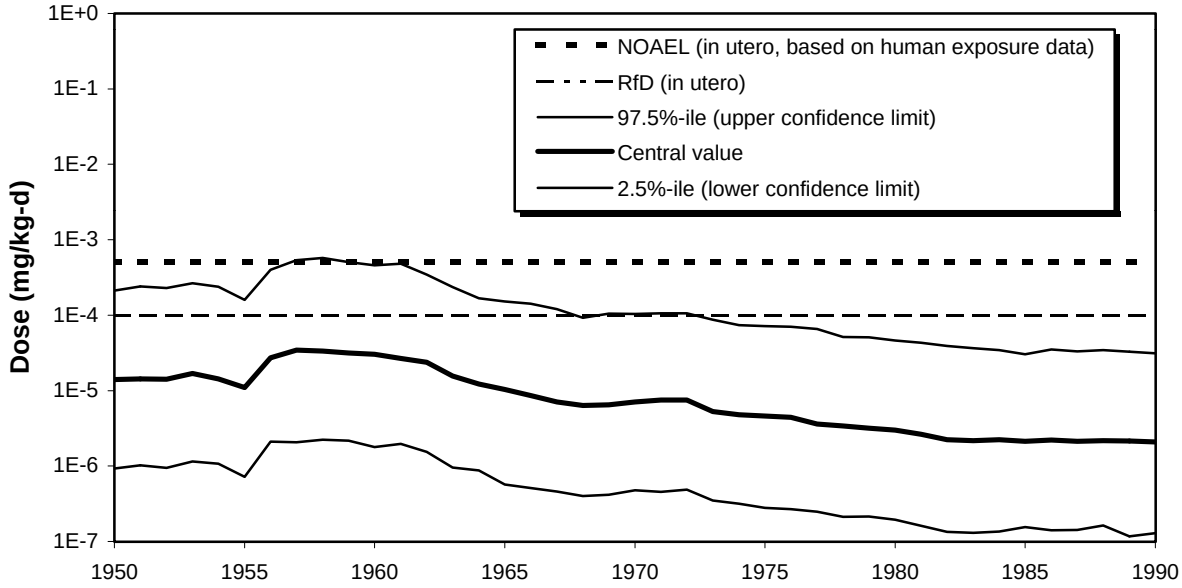


Figure 12-35: Clinch River/Poplar Creek Recreational Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

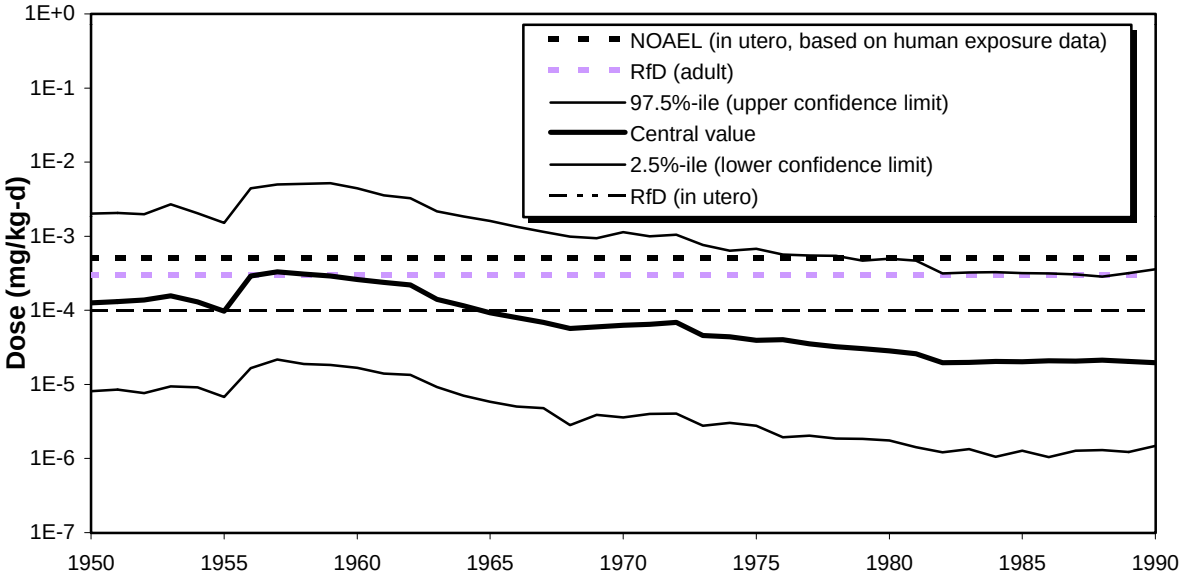


Figure 12-36: Clinch River/Poplar Creek Recreational Anglers-
Comparison of Estimated Methylmercury Doses (Child)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

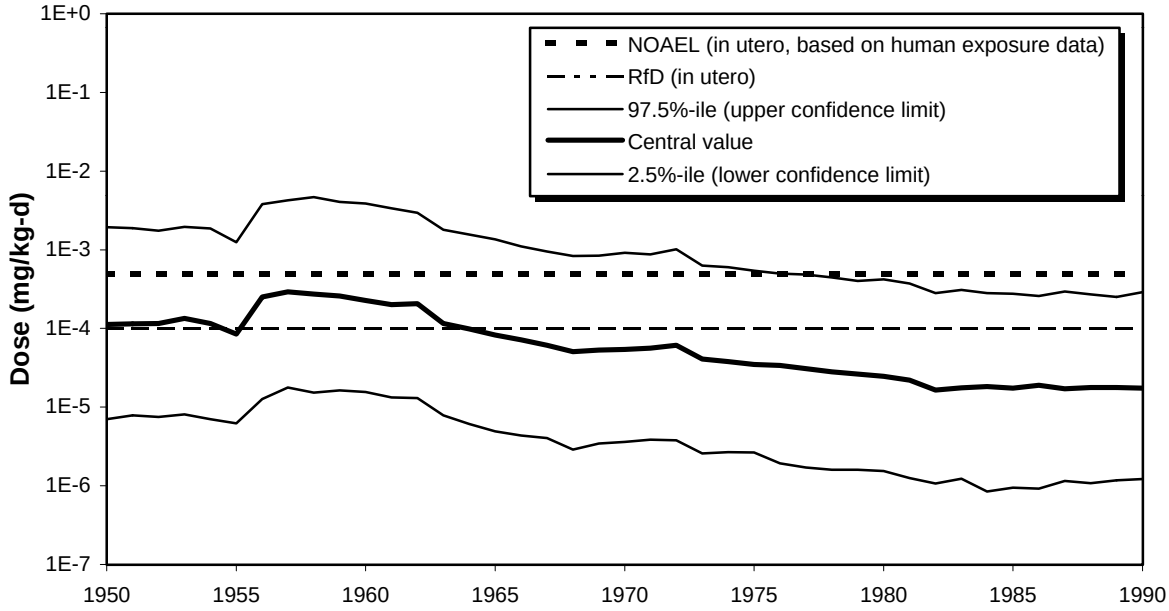
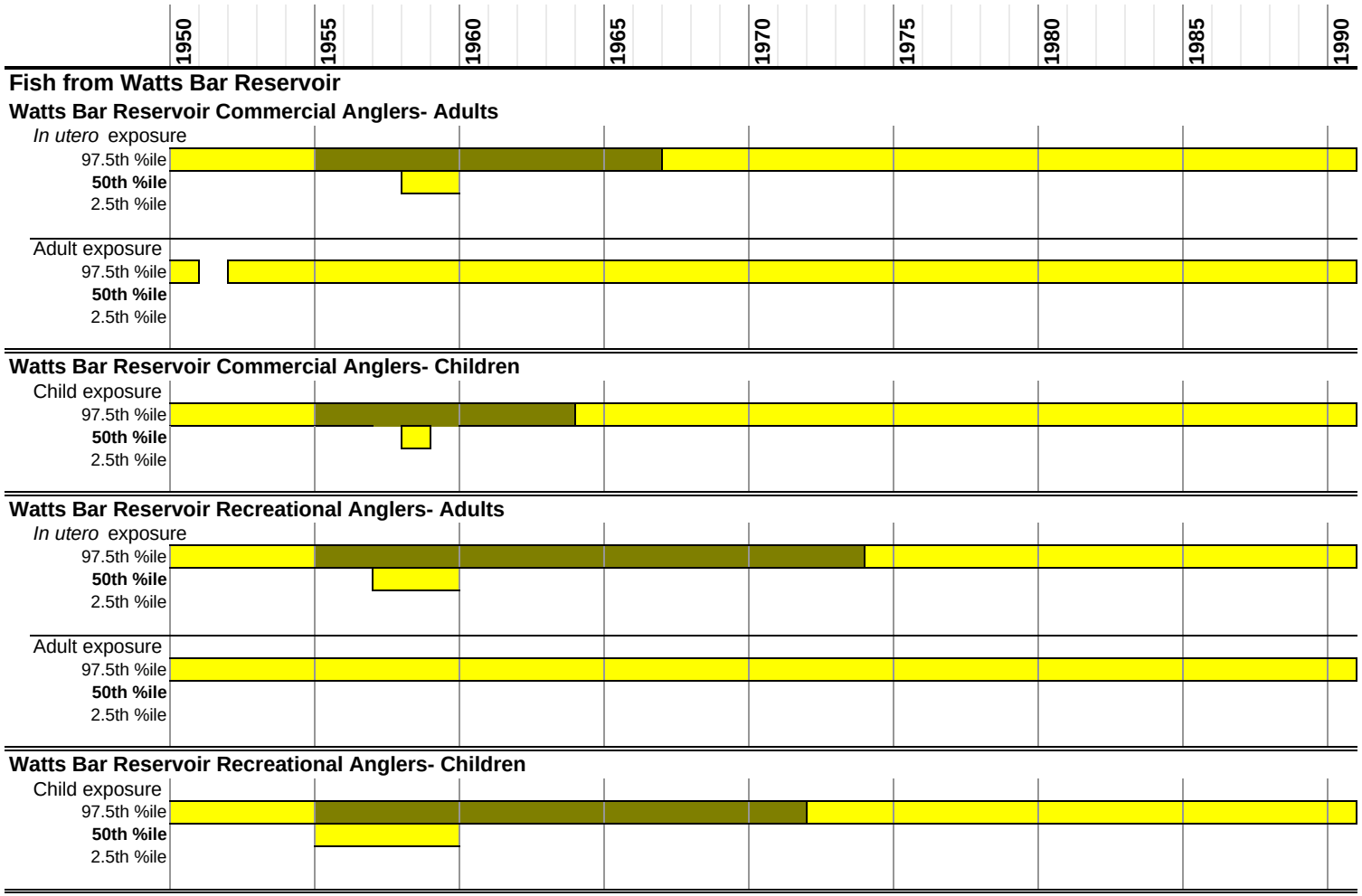
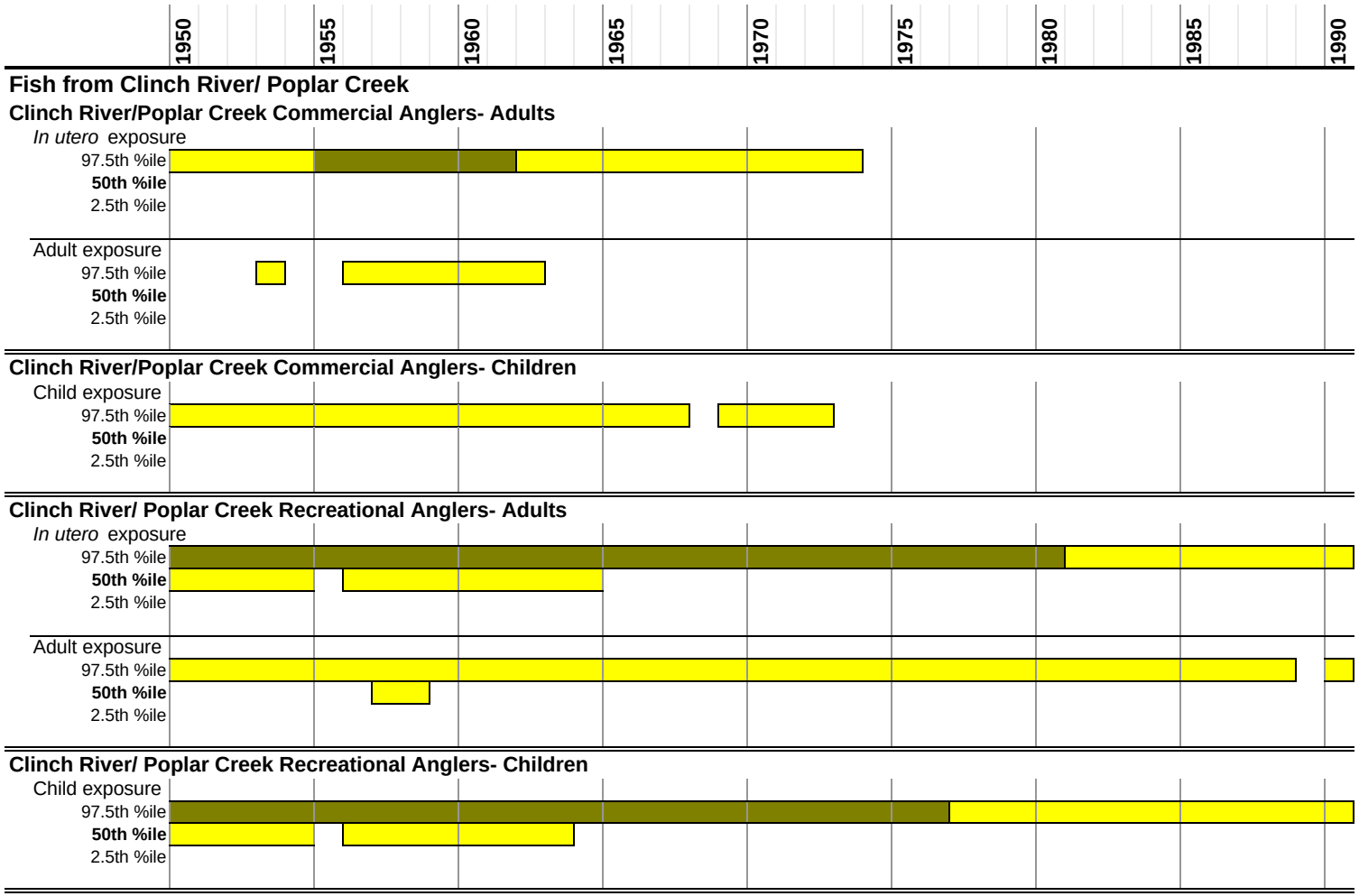


Figure 12-37: Years that the Estimated Mercury Doses from Consumption of Fish Exceeded the RfDs and the NOAEL-- Angler Populations [a]



a Years that exceeded the RfD are indicated with light shading (*in utero* RfD = 0.0001 mg kg^{-1} d^{-1}; adult RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0005 mg kg^{-1} d^{-1})

Figure 12-37: Years that the Estimated Mercury Doses from Consumption of Fish Exceeded the RfDs and the NOAEL-- Angler Populations [a]



a Years that exceeded the RfD are indicated with light shading (*in utero* RfD = 0.0001 mg kg^{-1} d^{-1}; adult RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0005 mg kg^{-1} d^{-1})

Table 12-15: Fish ConsumersS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	Estimated Population Size [a]	%-ile	Years Dose Exceeded the:			Ratio of Highest Dose (at given %-ile) to:		
			USEPA RfD [b]	Adult RfD [c]	NOAEL [d]	USEPA RfD [b]	Adult RfD [c]	NOAEL [d]
Watts Bar Reservoir								
Commercial Angler- Adult (compared to *in utero* and adult RfDs)								
Methyl-mercury	4-6/ 10-15	97.5%ile	1950-90	1950, 1952-90	1955-66	**19** *(1959)*	**6.3** *(1959)*	**3.8** *(1958)*
		50%ile	1958-59	---	---	**1.2** *(1958)*	0.40 *(1958)*	0.24 *(1959)*
		2.5%ile	---	---	---	0.072 *(1959)*	0.024 *(1959)*	0.014 *(1958)*
Commercial Angler- Child (compared to *in utero* RfD)								
Methyl-mercury	3-5	97.5%ile	1950-90	NA	1955-63	**16** *(1959)*	**NA**	**3.2** *(1958)*
		50%ile	1958	NA	---	**1.0** *(1958)*	NA	0.20 *(1959)*
		2.5%ile	---	NA	---	0.071 *(1958)*	NA	0.014 *(1959)*
Recreational Angler- Adult (compared to *in utero* and adult RfDs)								
Methyl-mercury	2,000-6,000/ 5,000-15,000	97.5%ile	1950-90	1950-90	1955-1973	**24** *(1959)*	**8.0** *(1959)*	**4.8** *(1958)*
		50%ile	1957-59	---	---	**1.5** *(1958)*	0.5 *(1958)*	0.38 *(1959)*
		2.5%ile	---	---	---	0.088 *(1958)*	0.029 *(1958)*	0.022 *(1958)*
Recreational Angler- Child (compared to *in utero* RfD)								
Methyl-mercury	1,500-4,500	97.5%ile	1950-90	NA	1955-71	**20** *(1959)*	NA	**4.0** *(1958)*
		50%ile	1957-59	NA	---	**1.3** *(1959)*	NA	0.26 *(1959)*
		2.5%ile	---	NA	---	0.083 *(1957)*	NA	0.017 *(1958)*

Table 12-15 (cont.): Fish ConsumersS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	Estimated Population Size [a]	%-ile	Years Dose Exceeded the:			Ratio of Highest Dose (at given %-ile) to:		
			USEPA RfD [b]	Adult RfD [c]	NOAEL [d]	USEPA RfD [b]	Adult RfD [c]	NOAEL [d]
Clinch River/ Poplar Creek								
Commercial Angler- Adult (compared to *in utero* and adult RfDs)								
Methyl-mercury	~1/ 1-3	97.5%ile	1950-73	1953, 1956-62	1956-61	**7.1** *(1957)*	**2.4** *(1957)*	**1.4** *(1957)*
		50%ile	---	---	---	0.39 *(1958)*	0.13 *(1958)*	0.078 *(1957)*
		2.5%ile	---	---	---	0.024 *(1958)*	0.008 *(1958)*	0.0048 *(1958)*
Commercial Angler- Child (compared to *in utero* RfD)								
Methyl-mercury	~1	97.5%ile	1950-67, 1969-72	NA	---	**5.7** *(1958)*	NA	**1.1** *(1957)*
		50%ile	---	NA	---	0.35 *(1957)*	NA	0.070 *(1957)*
		2.5%ile	---	NA	---	0.022 *(1958)*	NA	0.0044 *(1957)*
Recreational Angler- Adult (compared to *in utero* and adult RfDs)								
Methyl-mercury	600-2,000/ 1,500-5,000	97.5%ile	1950-90	1950-88, 1990	1950-80	**52** *(1959)*	**17** *(1959)*	**10** *(1957)*
		50%ile	1950-54, 1956-64	1957-58	---	**3.3** *(1957)*	**1.1** *(1957)*	0.66 *(1957)*
		2.5%ile	---	---	---	0.22 *(1957)*	0.073 *(1957)*	0.044 *(1957)*
Recreational Angler- Child (compared to *in utero* RfD)								
Methyl-mercury	450-1,500	97.5%ile	1950-90	NA	1950-76	**47** *(1958)*	NA	**9.4** *(1957)*
		50%ile	1950-54, 1956-63	NA	---	**2.9** *(1957)*	NA	0.58 *(1957)*
		2.5%ile	---	NA	---	0.18 *(1957)*	NA	0.036 *(1957)*

a Annual average population size; for adult exposure, the first range is for women of child-bearing age and the second range is for all adults

b USEPA RfD for methylmercury, based on *in utero* exposures = 0.0001 mg kg^{-1} d^{-1}

c RfD for adult exposures= 0.0003 mg kg^{-1} d^{-1}

c NOAEL for methylmercury, based on *in utero* exposures = 0.0005 mg kg^{-1} d^{-1}

For recreational and commercial anglers who ate fish from Watts Bar Reservoir, estimated mercury doses exceeded the methylmercury RfD (based on *in utero* exposures) for the following years:

- *Watts Bar Reservoir Commercial Anglers, Adults* **S** *Upper bounds* on the estimated mercury doses exceeded the RfD for all years; *central values* exceeded the RfD for 1958-1959. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1955-1966. *Central values* were less than 1/4 of the NOAEL.

- *Watts Bar Reservoir Commercial Anglers, Children* **S** *Upper bounds* on the estimated mercury doses exceeded the RfD for all years; *central values* exceeded the RfD for 1958. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1955-1963. *Central values* were less than 1/5 of the NOAEL.

- *Watts Bar Reservoir Recreational Anglers, Adults* **S** *Upper bounds* on the estimated mercury doses exceeded the RfD for all years; *central values* exceeded the RfD for 1957-1959. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1955-1973. *Central values* were less than ½ of the NOAEL.

- *Watts Bar Reservoir Recreational Anglers, Children* **S** *Upper bounds* on the estimated mercury doses exceeded the RfD for all years; *central values* exceeded the RfD for 1957-1959. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1955-1971. *Central values* were less than 1/3 of the NOAEL.

For recreational and commercial anglers who ate fish from Clinch River and/or Poplar Creek, the estimated mercury doses exceeded the USEPA methylmercury RfD (based on *in utero* exposures) for the following years:

- *Clinch River/Poplar Creek Commercial Anglers, Adults* **S** *Upper bounds* on the estimated mercury doses exceeded the RfD for 1950-1973. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1956-1961. *Central values* were less than 1/12 of the NOAEL.

- *Clinch River/Poplar Creek Commercial Anglers, Children* **S** *Upper bounds* on the estimated mercury doses exceeded the RfD for 1950-1967 and 1969-1972. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1957. *Central values* were less than 1/14 of the NOAEL.

- *Clinch River/Poplar Creek Recreational Anglers, Adults***S** *Upper bounds* on the estimated mercury doses exceeded the RfD for all years; *central values* exceeded the RfD for 1950-1954 and 1956-1964. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1950-1980. *Central values* were less than 2/3 of the NOAEL.

- *Clinch River/Poplar Creek Recreational Anglers, Children***S** *Upper bounds* on the estimated mercury doses exceeded the RfD for all years; *central values* exceeded the RfD for 1950-1954 and 1956-1963. *Upper bounds* exceeded the NOAEL (based on *in utero* exposures) for 1950-1976. *Central values* were less than 2/3 of the NOAEL.

Doses to Categories of Fish Consumers

Mercury doses were also estimated for three categories of fish consumers who ate different amounts of fish from each of these systems:

- *Category 1 Fish Consumers***S** individuals who ate more than 1 up to 2.5 fish meals per week (equivalent to approximately 24 to 61 g d^{-1}, assuming 170 g per fish meal, or 52 to 130 meals per year)

- *Category 2 Fish Consumers***S** individuals who ate more than 0.33 up to 1 fish meals per week (or more than 1 meal every three weeks to 1 meal per week, equivalent to approximately 8.0 to 24 g d^{-1}, assuming 170 g per fish meal, or 17 to 52 meals per year)

- *Category 3 Fish Consumers***S** individuals who ate between 0.04 and 0.33 fish meals per week (or 1 meal every six months to 1 meal every three weeks, equivalent to approximately 0.97 to 8.0 g d^{-1}, assuming 170 g per fish meal, or 2 to 17 meals per year).

Figure 12-38 shows how fish consumption rates for the three categories of fish consumers compare to the fish consumption rates assumed for the angler populations.

Figures 12-39 through 12-44 show the estimated annual average methylmercury doses for each category of fish consumer for Watts Bar Reservoir and Clinch River/Poplar Creek. Table 12-16 summarizes the years that the estimated doses to the three categories of fish consumers from consumption of fish exceeded the RfDs or the NOAEL. The highest doses were estimated to have been in 1957, 1958, and 1959, since these were the years that mercury releases to EFPC from Y-12 were estimated to be greatest.

Figure 12-38: Comparison of Fish Consumption Rates Assumed for Angler Populations and Categories of Fish Consumers

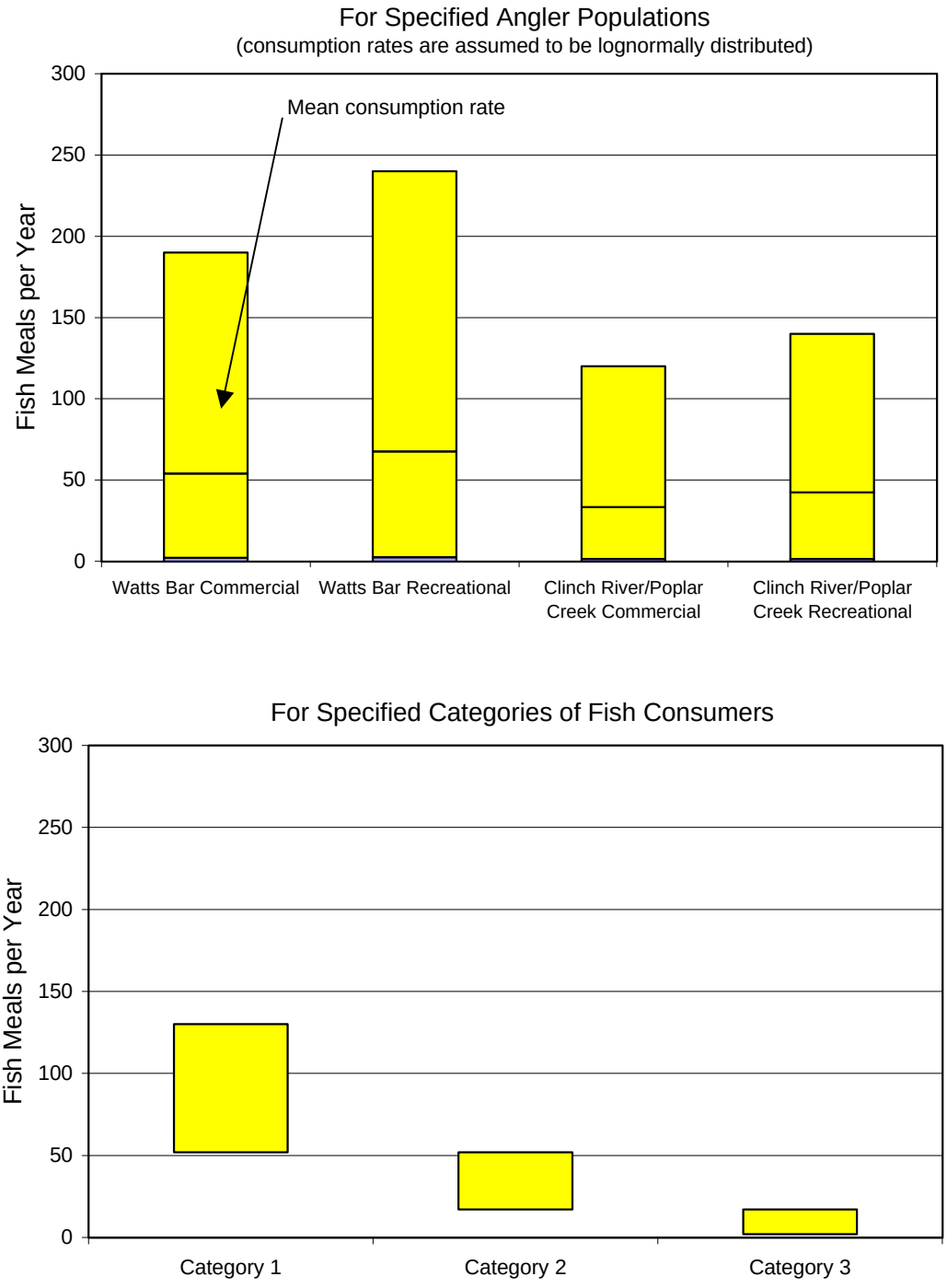


Figure 12-39: Watts Bar Reservoir Category 1 Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

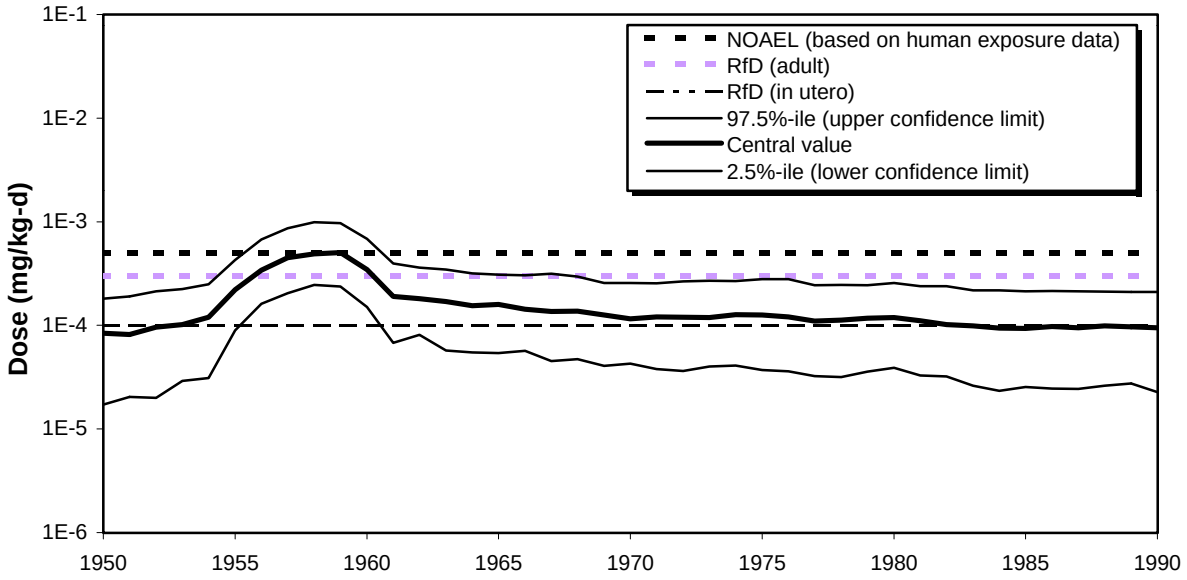


Figure 12-40: Clinch River/ Poplar Creek Category 1 Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

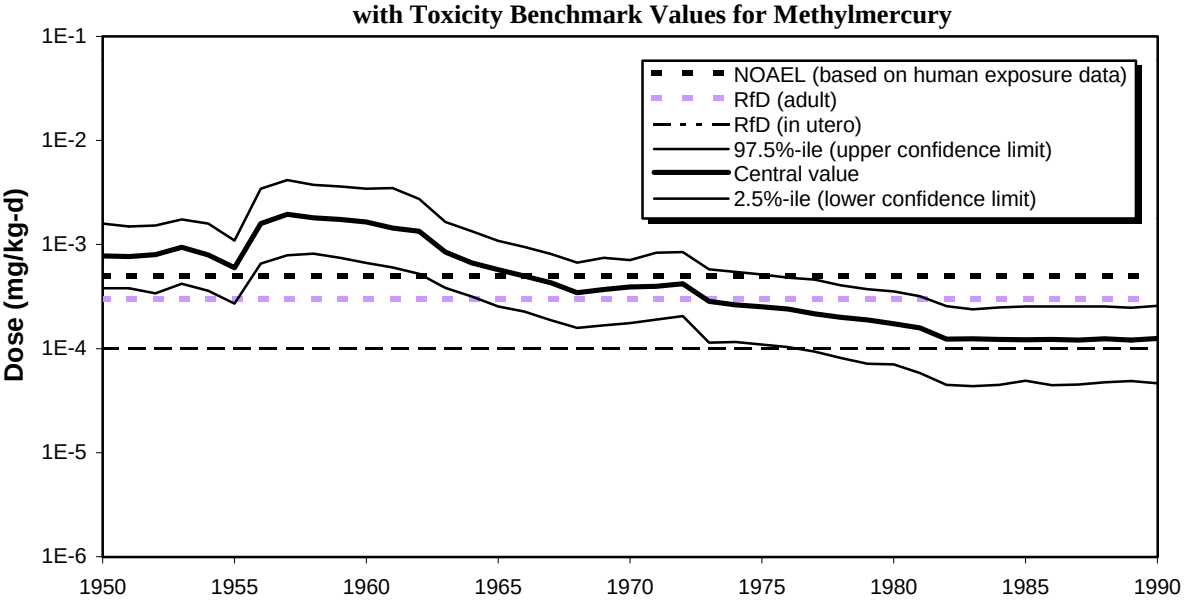


Figure 12-41: Watts Bar Reservoir Category 2 Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

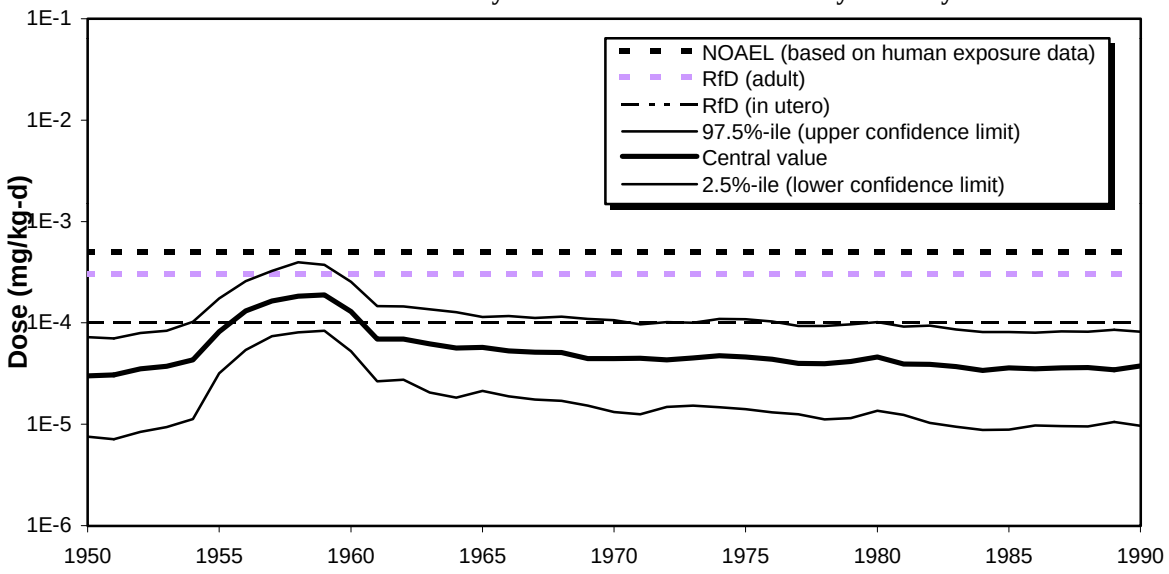


Figure 12-42: Clinch River/ Poplar Creek Category 2 Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

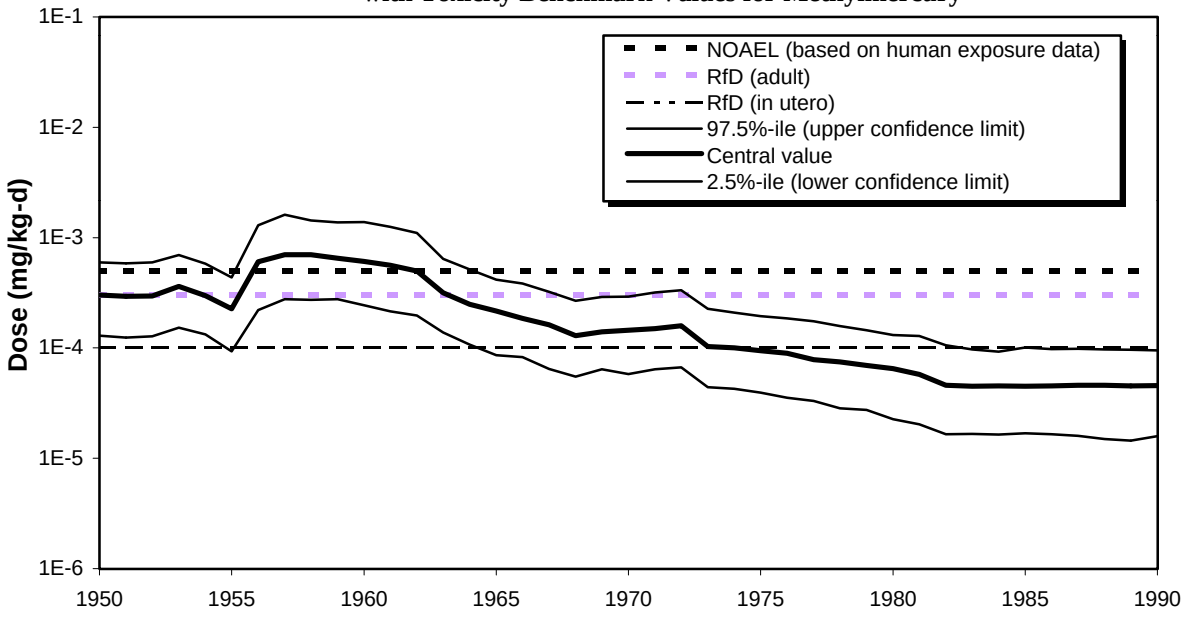


Figure 12-43: Watts Bar Reservoir Category 3 Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

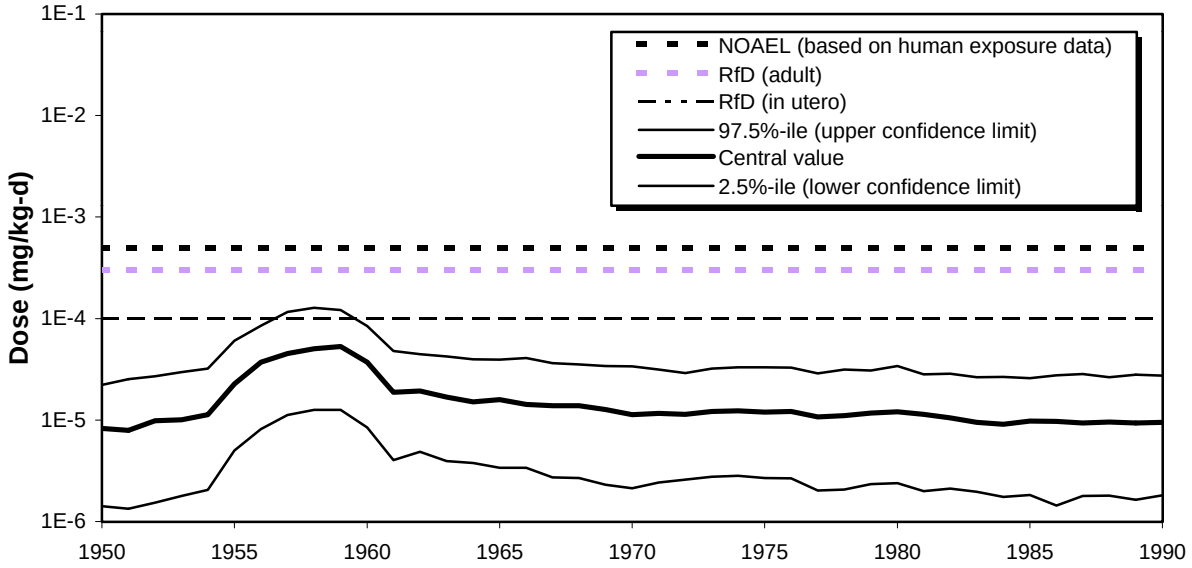


Figure 12-44: Clinch River/ Poplar Creek Category 3 Anglers-
Comparison of Estimated Methylmercury Doses (Adult)
from FISH CONSUMPTION
with Toxicity Benchmark Values for Methylmercury

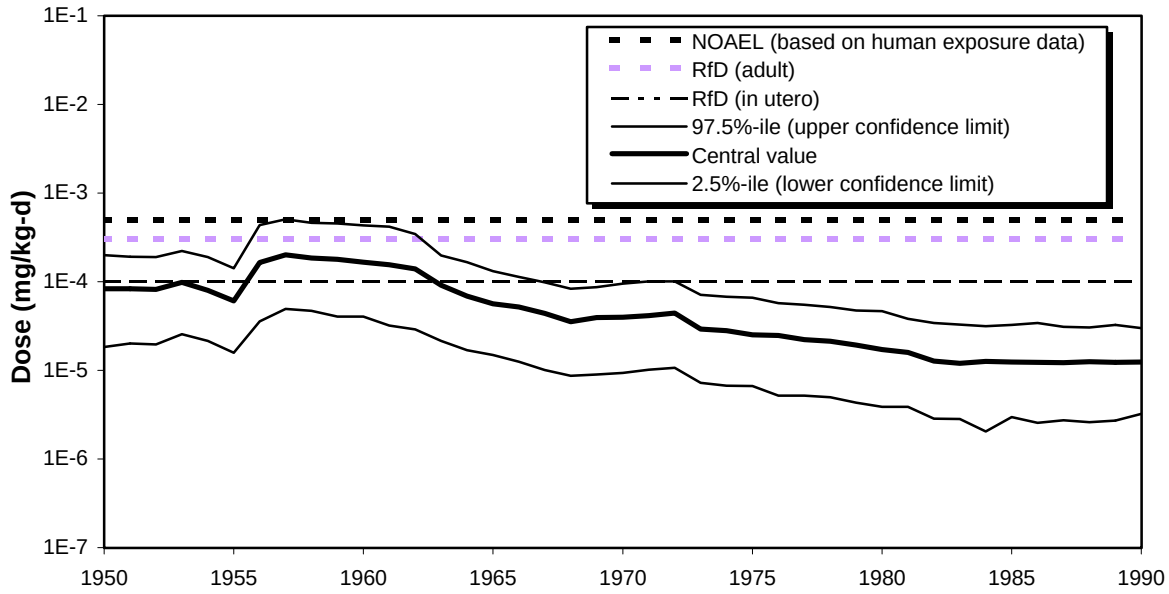


Table 12-16: Fish ConsumersS
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

Population	%-ile	Years Dose Exceeded the:			Ratio of Highest Dose (at given %-ile) to:		
		USEPA RfD[a]	Adult RfD[b]	NOAEL[c]	USEPA RfD[a]	Adult RfD[b]	NOAEL[c]
Watts Bar Reservoir							
Category 1- Adult							
Methylmercury	97.5%ile	1950-90	1950-80	1956-61	**9.9 *(1958)***	**3.3 *(1958)***	**2.0 *(1958)***
	50%ile	1950-82	1950-72	1959	**5.0 *(1959)***	**1.7 *(1959)***	**1.0 *(1959)***
	2.5%ile	1956-60	1950-54, 1956-64	---	**2.5 *(1958)***	0.83 *(1958)*	0.50 *(1958)*
Category 2- Adult							
Methylmercury	97.5%ile	1954-70, 1972-76, 1980	1954-70, 1972-76, 1980	---	**3.9 *(1958)***	**1.3 *(1958)***	0.78 *(1958)*
	50%ile	1956-60	1956-60	---	**1.9 *(1959)***	0.63 *(1959)*	0.38 *(1959)*
	2.5%ile	---	---	---	0.83 *(1959)*	0.28 *(1959)*	0.17 *(1959)*
Category 3- Adult							
Methylmercury	97.5%ile	1957-59	---	---	**1.3** *(1958)*	0.43 *(1958)*	0.26 *(1958)*
	50%ile	---	---	---	0.53 *(1959)*	0.18 *(1959)*	0.11 *(1959)*
	2.5%ile	---	---	---	0.13 *(1958)*	0.043 *(1958)*	0.026 *(1958)*
Clinch River/ Poplar Creek							
Category 1- Adult							
Methylmercury	97.5%ile	1950-90	1950-80	1950-75	**42 *(1957)***	**14 *(1957)***	**8.4 *(1957)***
	50%ile	1950-90	1950-72	1950-66	**20 *(1957)***	**6.7 *(1957)***	**4.0 *(1957)***
	2.5%ile	1950-75	1950-54, 1956-64	1957-62	**8.2 *(1958)***	**2.7 *(1958)***	**1.6 *(1958)***
Category 2- Adult							
Methylmercury	97.5%ile	1950-80	1950-67, 1971-73	1950-54, 1956-64	**16 *(1957)***	**5.3 *(1957)***	**3.2 *(1957)***
	50%ile	1950-72	1956-62	1956-62	**7.0 *(1957)***	**2.3 *(1957)***	**1.4 *(1957)***
	2.5%ile	1950-54, 1956-64	---	---	**2.8 *(1957)***	0.93 *(1957)*	0.56 *(1957)*
Category 3- Adult							
Methylmercury	97.5%ile	1950-66	1956-67	1957	**5.0 *(1957)***	**1.7 *(1957)***	**1.0 *(1957)***
	50%ile	1956-62	---	---	**2.0 *(1957)***	0.67 *(1957)*	0.40 *(1957)*
	2.5%ile	---	---	---	0.49 *(1957)*	0.16 *(1957)*	0.098 *(1957)*

a USEPA RfD for methylmercury, based on *in utero* exposures = 0.0001 mg kg^{-1} d^{-1}
b RfD for adult exposures= 0.0003 mg kg^{-1} d^{-1}; c NOAEL for methylmercury, based on *in utero* exposures = 0.0005 mg kg^{-1} d-1

Figure 12-45 shows how the mercury doses for all three categories of fish consumers who ate fish from Watts Bar Reservoir or Clinch River/Poplar Creek compare to the RfDs based on *in utero* and adult exposures and to the NOAEL at the 97.5th (upper), 50th (central), and 2.5th (lower) percentiles of the dose distributions.

For Category 1, 2, and 3 fish consumers that ate fish from Watts Bar Reservoir, the estimated mercury doses exceeded the methylmercury RfD (based on *in utero* exposures) for the following years:

- *Category 1 Fish Consumers***S** *Upper bounds* on the estimated mercury doses exceeded the RfD for all years; *central values* exceeded the RfD for 1953-1982. *Lower bounds* on the estimated mercury doses exceeded the RfD for 1956-1960.

- *Category 2 Fish Consumers***S** *Upper bounds* on the estimated mercury doses exceeded the RfD for 1954-1980; *central values* exceeded the RfD for 1956-1960. *Lower bounds* on the estimated mercury doses were below the RfD for all years.

- Category 3 Fish Consumers**S** *Upper bounds* on the estimated mercury doses exceeded the RfD for 1957-1959; *central values* were below the RfD for all years.

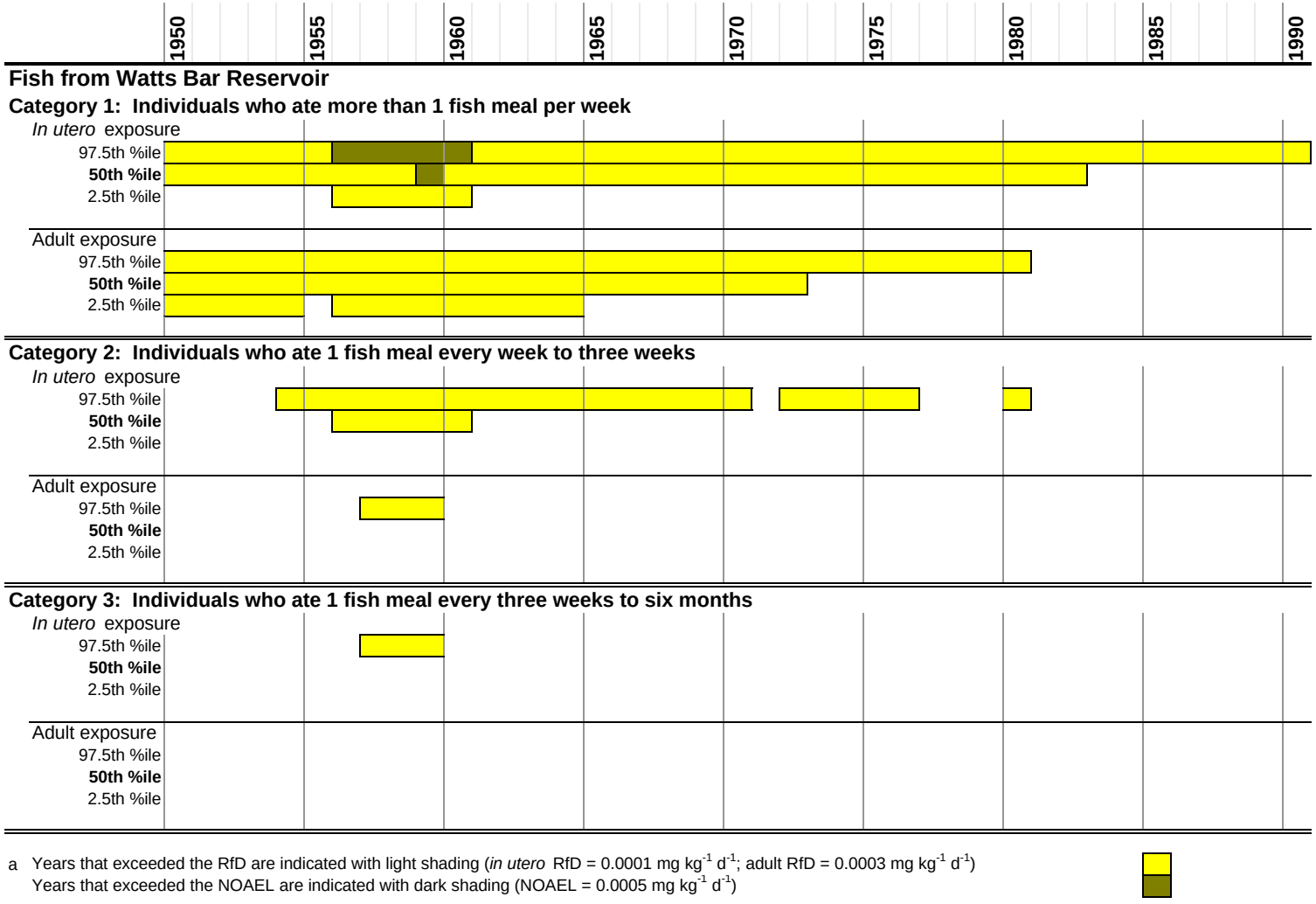
For Category 1, 2, and 3 fish consumers that ate fish from Clinch River and/or Poplar Creek, the estimated mercury doses exceeded the USEPA methylmercury RfD (0.0001 mg kg^{-1} d^{-1}, based on *in utero* exposures) for the following years:

- *Category 1 Fish Consumers***S** *Upper bounds* and *central values* on the estimated mercury doses exceeded the RfD for all years. *Lower bounds* on the estimated mercury doses exceeded the RfD for 1950-1976.

- *Category 2 Fish Consumers***S** *Upper bounds* on the estimated mercury doses exceeded the RfD for 1950-1985; *central values* exceeded the RfD for 1950-1974. *Lower bounds* on the estimated mercury doses exceeded the RfD for 1950-1964.

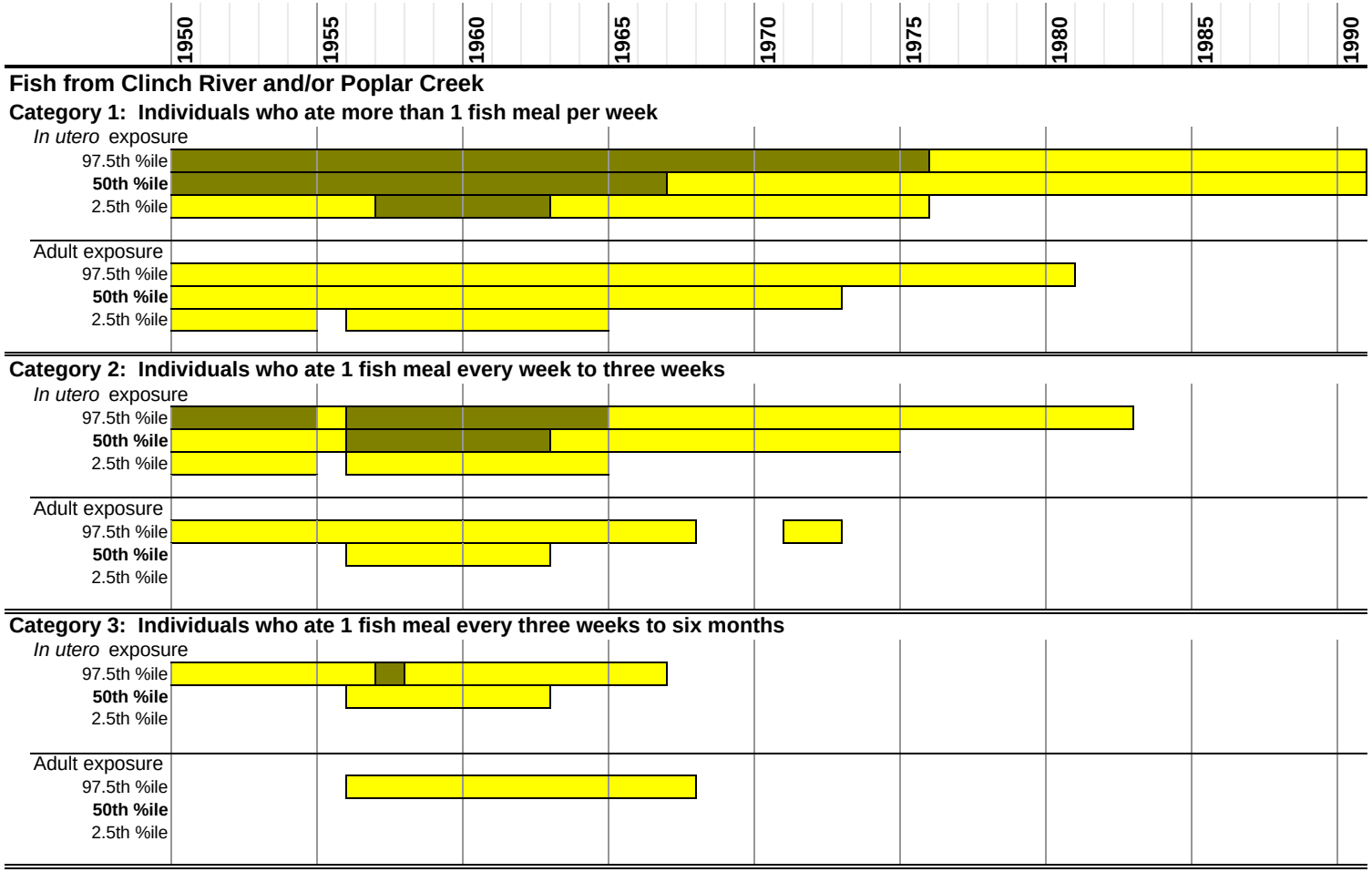
- *Category 3 Fish Consumers***S** *Upper bounds* on the estimated mercury doses exceeded the RfD for 1950-1966 and 1971-1972; *central values* exceeded the RfD for 1956-1962. *Lower bounds* were below the RfD for all years.

Figure 12-45: Years that the Estimated Mercury Doses from Consumption of Fish Exceeded the RfDs and the NOAEL-- Categories of Fish Consumers [a]



a Years that exceeded the RfD are indicated with light shading (*in utero* RfD = 0.0001 mg kg^{-1} d^{-1}; adult RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0005 mg kg^{-1} d^{-1})

Figure 12-45: Years that the Estimated Mercury Doses from Consumption of Fish Exceeded the RfDs and the NOAEL-- Categories of Fish Consumers [a]



a Years that exceeded the RfD are indicated with light shading (*in utero* RfD = 0.0001 mg kg^{-1} d^{-1}; adult RfD = 0.0003 mg kg^{-1} d^{-1})
Years that exceeded the NOAEL are indicated with dark shading (NOAEL = 0.0005 mg kg^{-1} d^{-1})

Methylmercury doses slightly above the NOAEL have been associated with observations of neurological effects in children who were exposed to methylmercury *in utero* (i.e., their mothers consumed methylmercury in fish).

Sensitivity analyses show that the dominant contributor to variance in estimated doses from fish consumption for the commercial and recreational angler scenarios is the consumption rate, followed by the concentration in fish (contributing 90% and 8.5%, respectively, of the total variance for the Watts Bar Commercial Angler/Adult during 1958). For the Category 1 and 2 fish consumers, the dominant contributors to variance were the fish concentration (contributing about 57-63%) and the fish consumption rate (contributing about 29-37%). For the Category 3 fish consumer, variance was dominated by the fish consumption rate (contributing about 61%), followed by the fish concentration (contributing about 35%). Detailed results of the sensitivity analysis are summarized in Appendix Y.

Tables 12-17 and 12-19 show the 95% subjective confidence interval on the number of meals of fish from Watts Bar Reservoir and Clinch River/Poplar Creek, respectively, that an adult female would have had to consume per year to have been exposed to methylmercury at an annual average dose equal to either the USEPA RfD for methylmercury based on *in utero* exposures or the RfD for methylmercury based on adult exposures. For purposes of this evaluation, the size of a fish meal for an adult female was assumed to range between 80 g and 250 g (a filet of fish about the size of a deck of cards weighs approximately 100 g). As shown, for 1957-1959, if a pregnant adult female consumed approximately 9 or more fish meals per year from Watts Bar Reservoir, she may have been exposed to methylmercury at a dose in excess of the USEPA RfD and her unborn child may have been at risk of postnatal neurological effects from *in utero* exposure. If a pregnant adult female consumed approximately 2 or more fish meals per year from Clinch River/Poplar Creek during 1953-1963, she may have been exposed to methylmercury at a dose in excess of the USEPA RfD.

Tables 12-18 and 12-20 show the 95% subjective confidence interval on the number of meals of fish from Watts Bar Reservoir and Clinch River/Poplar Creek, respectively, that a child would have had to consume per year to receive annual average methylmercury doses equal to the USEPA RfD for methylmercury based on *in utero* exposures. For purposes of this evaluation, the size of a fish meal for a child was assumed to range between 50 g and 120 g. As shown in these tables, if a small child consumed 3 to 4 meals of fish per year from Watts Bar Reservoir or 1 meal of fish per year from Clinch River/Poplar Creek between 1956 and 1960, there is a small probability that the child was exposed to methylmercury at an annual average dose equal to the USEPA RfD. If the child consumed about 7 meals of fish per year from Watts Bar Reservoir or 2 meals of fish from Clinch River/ Poplar Creek during these years, it is likely that the child was exposed to methylmercury at a dose in excess of the RfD. This RfD, however, is based on observations of neurological effects in children who were exposed *in utero*; it is not known whether very young children exposed to the same dose of methylmercury following birth are as susceptible to adverse health effects as if they were exposed *in utero*.

Table 12-17: 95% Subjective Confidence Intervals on the Meals of Fish from Watts Bar Reservoir Consumed per Year that Would have Yielded a Methylmercury Dose Equal to the RfD (based on *in utero* and adult exposures) [a, b]

Year	Adult (compared to *in utero* RfD)			Adult (compared to adult RfD)		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	49	120	480	150	350	1,400
1951	47	120	440	140	350	1,300
1952	37	98	470	110	290	1,400
1953	40	93	350	120	280	1,000
1954	35	82	290	110	250	880
1955	21	44	100	63	130	310
1956	14	28	72	41	84	220
1957	11	22	51	32	65	150
1958	9.0	19	43	26	57	130
1959	9.0	19	45	26	56	130
1960	13	28	68	40	83	210
1961	25	54	130	76	160	400
1962	24	50	130	72	150	400
1963	25	56	170	76	170	500
1964	29	61	200	87	180	590
1965	27	62	180	80	190	550
1966	31	66	200	92	200	600
1967	30	70	200	91	210	590
1968	30	67	180	90	200	550
1969	33	77	250	100	230	740
1970	35	80	240	100	240	730
1971	34	83	300	100	250	890
1972	36	83	300	110	250	890
1973	35	76	240	110	230	710
1974	33	78	240	100	230	720
1975	34	78	230	100	230	690
1976	33	82	260	100	250	770
1977	40	88	250	120	260	750
1978	36	85	270	110	250	800
1979	37	87	270	110	260	800
1980	33	81	230	100	240	680
1981	37	89	290	110	270	880
1982	38	86	360	120	260	1,100
1983	41	97	460	120	290	1,400
1984	43	110	420	130	320	1,300
1985	42	97	350	130	290	1,100
1986	40	96	460	120	290	1,400
1987	43	100	360	130	310	1,100
1988	42	96	360	120	290	1,100
1989	44	97	370	130	290	1,100
1990	44	99	430	130	300	1,300

a For the "*in utero* exposure" scenario, fish meal consumption rates equal to the RfD were calculated using the USEPA methylmercury RfD for *in utero* exposure of 0.0001 mg kg^{-1} d^{-1}. For the "adult exposure" scenario, fish meal consumption rates equal to the RfD were calculated using the adult exposure RfD of 0.0003 mg kg^{-1} d^{-1}.

b The 95%ile confidence intervals were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution, std dev = 9.5). 50^{th} %ile fish meal consumption rates for adult males equal to the RfD would likely be about 25% higher (than the values presented under "adult exposure", due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

Table 12-18: 95% Subjective Confidence Interval on the Meals of Fish from Watts Bar Reservoir Consumed per Year that Would have Yielded a Methylmercury Dose Equal to the RfD (based on *in utero* exposures) [a, b]

Year	Child (compared to *in utero* RfD)		
	2.5%ile	50%ile	97.5%ile
1950	20	43	190
1951	19	45	180
1952	16	37	140
1953	14	33	140
1954	14	29	93
1955	7.7	16	39
1956	5.0	9.9	22
1957	3.8	7.6	18
1958	3.6	7.1	16
1959	3.4	7.1	16
1960	5.0	10	22
1961	8.7	19	47
1962	8.9	18	54
1963	9.2	21	55
1964	11	23	63
1965	11	23	57
1966	12	24	68
1967	12	26	75
1968	11	26	80
1969	12	29	89
1970	12	30	96
1971	14	29	97
1972	13	29	95
1973	13	29	93
1974	13	29	88
1975	12	28	84
1976	13	30	98
1977	15	31	120
1978	13	32	120
1979	14	29	96
1980	14	30	98
1981	15	32	110
1982	14	33	130
1983	15	35	130
1984	15	37	140
1985	16	36	160
1986	16	36	180
1987	17	38	130
1988	16	35	110
1989	15	35	150
1990	16	38	120

a Child fish meal consumption rates equal to the RfD were calculated using the USEPA methylmercury RfD for *in utero* exposure of 0.0001 mg kg^{-1} d^{-1}.
b The 95%ile confidence intervals were calculated using body weights representative of children (age 6 mo-3 yrs), assumed to have an average body weight of 12 kg (normal distribution, std dev = 2.2).

Table 12-19: 95% Subjective Confidence Intervals on the Meals of Fish from Clinch River/Poplar Creek Consumed per Year that Would have Yielded a Methylmercury DoseEqual to the RfD (based on *in utero* and adult exposure) [a, b]

Year	Adult (compared to *in utero* RfD)			Adult (compared to adult RfD)		
	2.5%ile	50%ile	97.5%ile	2.5%ile	50%ile	97.5%ile
1950	5.9	12	29	18	35	86
1951	5.9	12	27	18	36	82
1952	5.9	11	26	18	34	78
1953	4.7	10	24	14	30	73
1954	6.1	12	30	18	36	91
1955	8.1	16	38	24	47	115
1956	2.8	6.0	15	8.4	18	45
1957	2.2	5.0	12	6.6	14	36
1958	2.2	5.0	13	6.5	15	40
1959	2.6	5.0	13	7.7	16	39
1960	2.6	6.0	16	7.8	18	47
1961	3.0	6.0	18	9.0	19	53
1962	3.0	7.0	19	8.9	21	56
1963	6.0	11	26	18	34	78
1964	7.1	14	30	21	42	91
1965	8.1	17	43	24	51	130
1966	10	19	49	29	58	150
1967	10	22	56	31	66	170
1968	13	26	57	39	78	170
1969	12	26	64	36	78	190
1970	12	24	58	36	73	170
1971	12	24	63	35	72	190
1972	11	24	56	32	71	170
1973	16	32	77	48	95	230
1974	17	34	94	50	100	280
1975	17	37	88	52	110	270
1976	19	39	97	56	120	290
1977	21	44	110	62	130	330
1978	23	47	120	68	140	370
1979	25	51	130	75	150	400
1980	26	57	140	78	170	410
1981	28	66	170	85	200	520
1982	35	75	220	100	230	660
1983	35	79	220	100	240	650
1984	37	76	230	110	230	700
1985	33	78	230	100	230	700
1986	36	78	260	110	240	780
1987	36	80	240	110	240	720
1988	35	77	200	110	230	590
1989	35	79	240	110	240	710
1990	37	80	220	110	240	670

a For the "*in utero* exposure" scenario, fish meal consumption rates equal to the RfD were calculated using the USEPA methylmercury RfD for *in utero* exposure of 0.0001 mg kg^{-1} d^{-1}. For the "adult exposure" scenario, fish meal consumption rates equal to the RfD were calculated using the adult exposure RfD of 0.0003 mg kg^{-1} d^{-1}.

b The 95%ile confidence intervals were calculated using body weights representative of adult females, assumed to have an average body weight of 62 kg (lognormal distribution, std dev = 9.5). 50^{th} %ile fish meal consumption rates for adult males equal to the RfD would likely be about 25% higher (than the values presented under "adult exposure", due to their higher average body weight (the average body weight of an adult male is about 78 kg, USEPA 1995).

Table 12-20: 95% Subjective Confidence Intervals on the Meals of Fish from Clinch River/Poplar Creek Consumed per Year that Would have Yielded a Methylmercury Dose Equal to the RfD (based on *in utero* RfD) [a, b]

Year	Child (compared to *in utero* RfD)		
	2.5%ile	50%ile	97.5%ile
1950	2.2	4.4	10
1951	2.1	4.6	9.1
1952	2.2	4.4	10
1953	1.7	3.6	8.2
1954	2.2	4.4	9.2
1955	2.9	6.0	14
1956	1.0	2.1	5.2
1957	0.8	1.7	4.3
1958	0.9	2.0	4.4
1959	0.9	2.0	5.1
1960	1.0	2.1	5.1
1961	1.1	2.4	6.2
1962	1.1	2.5	6.2
1963	2.0	4.1	8.8
1964	2.5	5.5	11
1965	3.2	6.2	13
1966	3.6	7.0	16
1967	4.1	8.4	19
1968	4.5	10	21
1969	5.0	9.3	19
1970	4.5	9.0	21
1971	4.3	8.6	18
1972	4.0	8.3	18
1973	5.6	13	29
1974	5.9	13	30
1975	6.8	14	34
1976	6.7	15	33
1977	7.4	16	36
1978	8.4	18	43
1979	8.8	18	46
1980	9.4	21	58
1981	11	23	61
1982	13	28	71
1983	13	29	72
1984	13	29	82
1985	12	29	78
1986	13	30	73
1987	13	28	69
1988	13	29	81
1989	12	29	75
1990	14	29	70

a Child fish meal consumption rates equal to the RfD were calculated using the USEPA methylmercury RfD for *in utero* exposure of 0.0001 mg kg^{-1} d^{-1}.
b The 95%ile confidence intervals were calculated using body weights representative of children (age 6 mo-3 yrs), assumed to have an average body weight of 12 kg (normal distribution, std dev = 2.2).

In a given year, some individuals may have consumed fish from more than one river system, for example Watts Bar Reservoir, Clinch River/Poplar Creek, and EFPC. Table 12-21 shows the mean annual average methylmercury dose per meal of fish consumed (mg kg^{-1} d^{-1} per meal) from Watts Bar Reservoir, Clinch River/Poplar Creek, and EFPC. The dose per meal can be multiplied by the number of meals consumed per year from each system to derive an estimate of the annual average mercury dose due to consumption of fish from that system. Annual average doses for each system can then be added to derive an estimate of an individual's total annual average mercury dose from fish consumption.

Population Size

As discussed in Section 6.2, the estimated total size of the angler populations evaluated in this assessment was assumed to be as follows:

- *Watts Bar Commercial Anglers***S** this population was assumed to be small, with a total during a given year of 20 to 30 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 10 to 15 individuals), adult females of child-bearing age comprised approximately 20% of the population (or about 4 to 6 individuals), and children ages 6 months to 3 years comprised approximately 15% of the population (or about 3 to 5 individuals).

- *Watts Bar Recreational Anglers***S** this population was assumed to be very large, with a total during a given year of 10,000 to 30,000 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 5,000 to 15,000 individuals), adult females of child-bearing age comprised approximately 20% of the population (or about 2,000 to 6,000 individuals), and children ages 6 months to 3 years comprised approximately 15% of the population (or about 1,500 to 4,500 individuals).

- *Clinch River/Poplar Creek Commercial Anglers***S** this population was assumed to very small, with a total during a given year of 1 to 5 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 1 to 3 individuals), adult females of child-bearing age comprised approximately 20% of the population (or one individual), and children ages 6 months to 3 years comprised approximately 15% of the population (or one individual).

T able 12-21: Mean Annual Average Methylmercury Dose per Fish Meal Consumed (Adult Female, mg kg^{-1} d^{-1})

Year	Fish from Watts Bar Reservoir		Fish from Clinch River/ Poplar Creek		Fish from EFPC	
	Mean	SD[a]	Mean	SD	Mean	SD
1950	9.3×10^{-7}	4.8×10^{-7}	8.7×10^{-6}	3.4×10^{-6}	2.0×10^{-5}	7.9×10^{-6}
1951	9.8×10^{-7}	5.0×10^{-7}	9.1×10^{-6}	3.6×10^{-6}	2.1×10^{-5}	8.3×10^{-6}
1952	1.1×10^{-6}	5.6×10^{-7}	8.8×10^{-6}	3.5×10^{-6}	2.0×10^{-5}	7.9×10^{-6}
1953	1.2×10^{-6}	5.9×10^{-7}	1.1×10^{-5}	4.2×10^{-6}	2.1×10^{-5}	7.9×10^{-6}
1954	1.3×10^{-6}	6.4×10^{-7}	8.7×10^{-6}	3.5×10^{-6}	2.0×10^{-5}	8.0×10^{-6}
1955	2.4×10^{-6}	1.0×10^{-6}	6.7×10^{-6}	2.7×10^{-6}	2.0×10^{-5}	8.2×10^{-6}
1956	3.8×10^{-6}	1.6×10^{-6}	1.8×10^{-5}	8.2×10^{-6}	2.0×10^{-5}	8.2×10^{-6}
1957	4.9×10^{-6}	2.0×10^{-6}	2.2×10^{-5}	9.5×10^{-6}	2.0×10^{-5}	8.2×10^{-6}
1958	5.7×10^{-6}	2.2×10^{-6}	2.1×10^{-5}	9.5×10^{-6}	2.1×10^{-5}	8.7×10^{-6}
1959	5.7×10^{-6}	2.3×10^{-6}	2.0×10^{-5}	8.8×10^{-6}	2.0×10^{-5}	8.5×10^{-6}
1960	4.0×10^{-6}	1.7×10^{-6}	1.9×10^{-5}	8.5×10^{-6}	2.1×10^{-5}	8.7×10^{-6}
1961	2.1×10^{-6}	8.9×10^{-7}	1.7×10^{-5}	7.8×10^{-6}	2.0×10^{-5}	8.0×10^{-6}
1962	2.2×10^{-6}	9.5×10^{-7}	1.6×10^{-5}	6.9×10^{-6}	2.1×10^{-5}	8.5×10^{-6}
1963	1.8×10^{-6}	7.6×10^{-7}	9.2×10^{-6}	3.4×10^{-6}	2.0×10^{-5}	7.7×10^{-6}
1964	1.8×10^{-6}	7.7×10^{-7}	7.8×10^{-6}	3.0×10^{-6}	2.1×10^{-5}	7.8×10^{-6}
1965	1.8×10^{-6}	7.6×10^{-7}	6.5×10^{-6}	2.6×10^{-6}	2.0×10^{-5}	7.1×10^{-6}
1966	1.7×10^{-6}	7.4×10^{-7}	5.6×10^{-6}	2.3×10^{-6}	2.0×10^{-5}	7.4×10^{-6}
1967	1.6×10^{-6}	7.3×10^{-7}	4.9×10^{-6}	2.2×10^{-6}	2.0×10^{-5}	8.0×10^{-6}
1968	1.6×10^{-6}	7.3×10^{-7}	3.9×10^{-6}	1.6×10^{-6}	1.9×10^{-5}	7.1×10^{-6}
1969	1.4×10^{-6}	6.7×10^{-7}	4.3×10^{-6}	1.8×10^{-6}	1.9×10^{-5}	7.2×10^{-6}
1970	1.4×10^{-6}	6.6×10^{-7}	4.5×10^{-6}	1.8×10^{-6}	2.0×10^{-5}	7.9×10^{-6}
1971	1.3×10^{-6}	6.7×10^{-7}	4.6×10^{-6}	1.9×10^{-6}	1.9×10^{-5}	7.4×10^{-6}
1972	1.3×10^{-6}	6.3×10^{-7}	4.6×10^{-6}	1.9×10^{-6}	1.8×10^{-5}	6.9×10^{-6}
1973	1.4×10^{-6}	6.3×10^{-7}	3.2×10^{-6}	1.3×10^{-6}	1.9×10^{-5}	7.1×10^{-6}
1974	1.5×10^{-6}	7.0×10^{-7}	3.1×10^{-6}	1.3×10^{-6}	1.9×10^{-5}	7.2×10^{-6}
1975	1.5×10^{-6}	6.8×10^{-7}	2.8×10^{-6}	1.2×10^{-6}	1.7×10^{-5}	6.0×10^{-6}
1976	1.3×10^{-6}	6.5×10^{-7}	2.7×10^{-6}	1.1×10^{-6}	1.6×10^{-5}	6.0×10^{-6}
1977	1.3×10^{-6}	6.0×10^{-7}	2.5×10^{-6}	1.0×10^{-6}	1.6×10^{-5}	5.6×10^{-6}
1978	1.3×10^{-6}	6.2×10^{-7}	2.4×10^{-6}	1.1×10^{-6}	1.6×10^{-5}	5.9×10^{-6}
1979	1.3×10^{-6}	6.2×10^{-7}	2.2×10^{-6}	9.4×10^{-7}	1.5×10^{-5}	5.4×10^{-6}
1980	1.4×10^{-6}	6.5×10^{-7}	1.9×10^{-6}	8.9×10^{-7}	1.4×10^{-5}	5.4×10^{-6}
1981	1.3×10^{-6}	6.3×10^{-7}	1.7×10^{-6}	7.6×10^{-7}	1.4×10^{-5}	5.1×10^{-6}
1982	1.2×10^{-6}	5.8×10^{-7}	1.4×10^{-6}	6.6×10^{-7}	1.3×10^{-5}	5.0×10^{-6}
1983	1.2×10^{-6}	5.6×10^{-7}	1.4×10^{-6}	6.2×10^{-7}	1.2×10^{-5}	4.0×10^{-6}
1984	1.0×10^{-6}	5.3×10^{-7}	1.4×10^{-6}	6.5×10^{-7}	1.0×10^{-5}	3.8×10^{-6}
1985	1.1×10^{-6}	5.5×10^{-7}	1.4×10^{-6}	6.6×10^{-7}	7.6×10^{-6}	2.8×10^{-6}
1986	1.1×10^{-6}	5.5×10^{-7}	1.4×10^{-6}	6.3×10^{-7}	8.0×10^{-6}	2.6×10^{-6}
1987	1.0×10^{-6}	5.8×10^{-7}	1.3×10^{-6}	5.7×10^{-7}	7.5×10^{-6}	2.6×10^{-6}
1988	1.1×10^{-6}	5.8×10^{-7}	1.3×10^{-6}	6.0×10^{-7}	7.5×10^{-6}	2.6×10^{-6}
1989	1.0×10^{-6}	5.7×10^{-7}	1.3×10^{-6}	5.9×10^{-7}	7.4×10^{-6}	2.6×10^{-6}
1990	1.1×10^{-6}	5.6×10^{-7}	1.4×10^{-6}	6.3×10^{-7}	7.5×10^{-6}	2.6×10^{-6}

SD = Standard deviation

- *Clinch River/Poplar Creek Recreational Anglers***S** this population was assumed to be large, with a total during a given year of 3,000 to 10,000 people. Of these, it was estimated that adults (males and females) comprised approximately 50% of the total population (or about 1,500 to 5,000 individuals), adult females of child-bearing age comprised approximately 20% of the population (or about 600 to 2,000 individuals), and children ages 6 months to 3 years comprised approximately 15% of the population (or about 450 to 1,500 individuals).

- *Number of fetuses potentially affected*– The number of fetuses that may have been affected (average doses greater than the NOAEL) was computed using the average birth rate in the population, the fraction of women of childbearing age in the population, their fish consumption rates, and the fraction of consumers whose doses exceeded the NOAEL for *in utero* exposure during each year.[1] The estimates were made for Watts Bar Reservoir, Clinch River/Poplar Creek, and EFPC fish consumers and summed over the years of concern. The estimated number of fetuses placed at risk is uncertain, but is nearer to 100 than 1,000.

Conclusions

Based on this evaluation, mercury doses to young children or pregnant adult females who consumed fish from the Clinch River and/or Poplar Creek may have exceeded the USEPA RfD (based on neurological effects following *in utero* exposures) if, during the years of highest mercury releases from Y-12 (mid-1950s to mid-1960s), these individuals ate as few as 1 to 2 fish meals per year from these systems. If, during these years, a young child ate 2 to 3 or more meals per year, or a pregnant adult female ate about 5 or more meals per year from these systems, it is *likely* that their doses exceeded the USEPA RfD for methylmercury.

Mercury doses to people who consumed fish from Watts Bar Reservoir were likely lower per meal of fish consumed than doses associated with consumption of fish from Clinch River/ Poplar Creek (the dose per meal of Watts Bar Reservoir fish was likely about 1/4 of the dose per meal of Clinch River/ Poplar Creek fish). Consequently, if a young child ate 3 to 4 meals of fish per year or a pregnant adult female ate about 10 or more meals of fish per year from Watts Bar Reservoir during the years of highest mercury releases from Y-12, their doses may have exceeded the USEPA RfD. If a young child ate about 7 meals of fish per year, or a pregnant adult female ate about 20 meals of fish per year from Watts Bar Reservoir in these years, their doses *likely* exceeded the USEPA RfD.

In this assessment, the adult RfD for methylmercury is assumed to be three times larger than the USEPA RfD protective of *in utero* exposures. Consequently, it is likely that non-pregnant adult females and other adults could have consumed about three times more fish from these systems per year as pregnant adult females to have been exposed to methylmercury at doses equal to the adult RfD instead of the *in utero*

[1] Personal communication with Paul Voillequé of the ORHASP. June 1999.

RfD. During the highest release years, non-pregnant adults who consumed about 15 or more fish meals per year from Clinch River/ Poplar Creek or about 60 or more fish meals per year from Watts Bar Reservoir were *likely* exposed to mercury above the adult RfD.

It is likely that the majority of fish meals consumed from Clinch River/ Poplar Creek or Watts Bar Reservoir were consumed by adults who were not pregnant. The numbers of individuals who consumed fish from either Clinch River/Poplar Creek or Watts Bar Reservoir was likely large, however, and a significant number of these individuals were likely to have been young children or pregnant adult females. Further, because of the large number of people likely to have consumed fish from these systems, and the likelihood that some of these individuals were avid anglers and consumed large amounts of fish from these systems, it is likely that some non-pregnant adult females and other adults who consumed fish from these systems were exposed to mercury at doses that exceeded the *adult* RfD.

12.2 Observations and Conclusions

The following general conclusions can be drawn from the Task 2 mercury dose reconstruction:

Inhalation of airborne (elemental) mercury:

(Figures 12-46 and 12-47 show how the highest estimated elemental mercury doses for adults and children, respectively, for each population compare to the RfD and the NOAEL)

- *Comparison to RfDs***S**The 95% confidence interval on the estimated inhalation doses of elemental mercury exceeded the RfD at two population locations: the *Scarboro Community* in 1955, 1957, and 1958 (child) and the *EFPC Floodplain Farm Family* location for 1953-1960 (child) and 1955-1959 (adult). At both of these locations, estimated doses exceeded the RfD at the *upper bound* of the distribution (the 97.5th percentile). At the *EFPC Floodplain Farm Family* location, *central values* (50th percentiles) for the child also exceeded the RfD for 1955 and 1957-1958.

- *Comparison to NOAELs***S** The 95% subjective confidence intervals on the estimated annual average elemental mercury doses for *all* populations and *all* years were below the NOAELs. These NOAELs were established from studies of workers exposed to airborne mercury vapor for prolonged periods of time. Neurological effects, including hand tremor, increases in memory disturbances, and evidence of dysfunction of the autonomic (involuntary) nervous system (IRIS 1998) were reported in some workers exposed at doses above the NOAELs. At slightly higher doses, evidence of effects on the kidney have also been observed. The USEPA RfD is about 30 times lower than the NOAEL because it incorporates a conservative safety factor. Health effects in humans exposed to elemental mercury at doses at or below the NOAEL have not been reported.

Figure 12-46: Elemental Mercury (Adult exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values

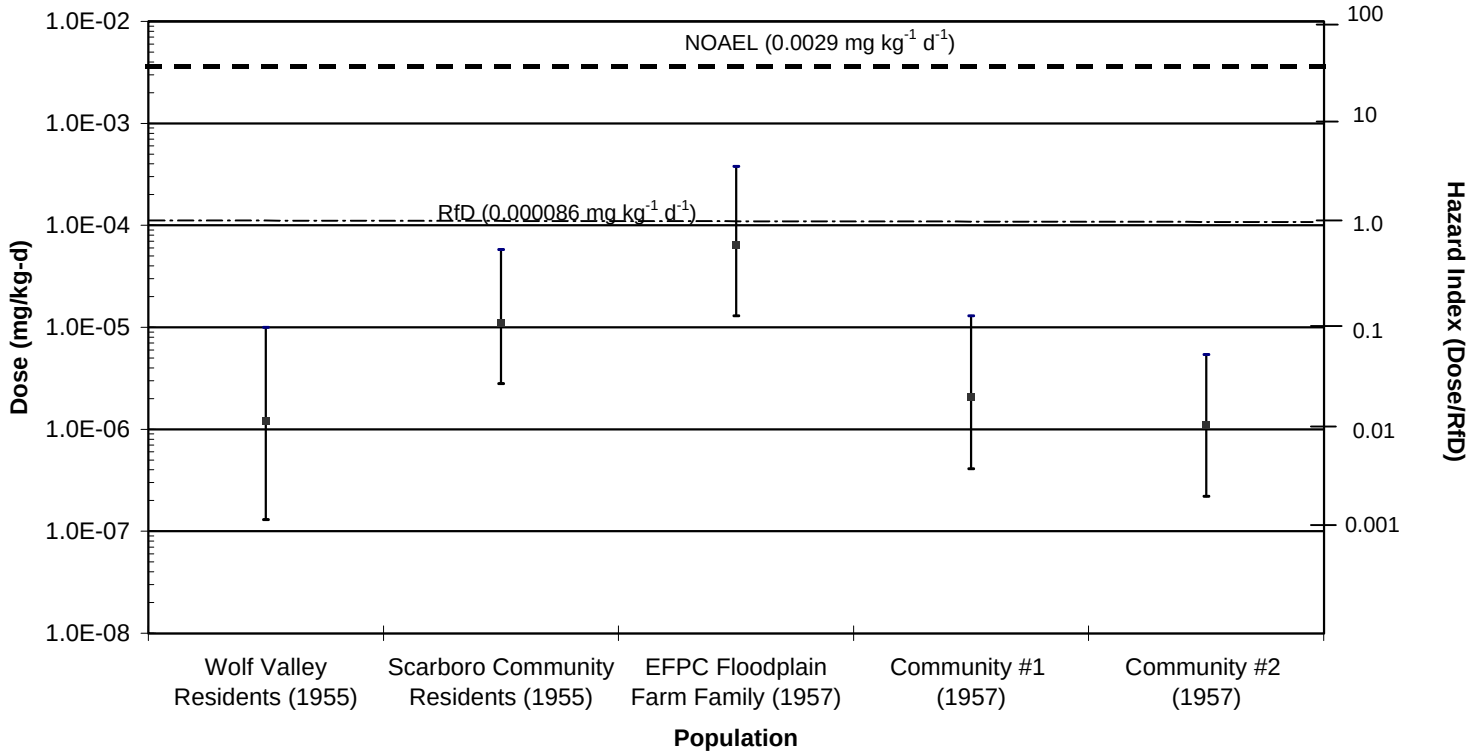
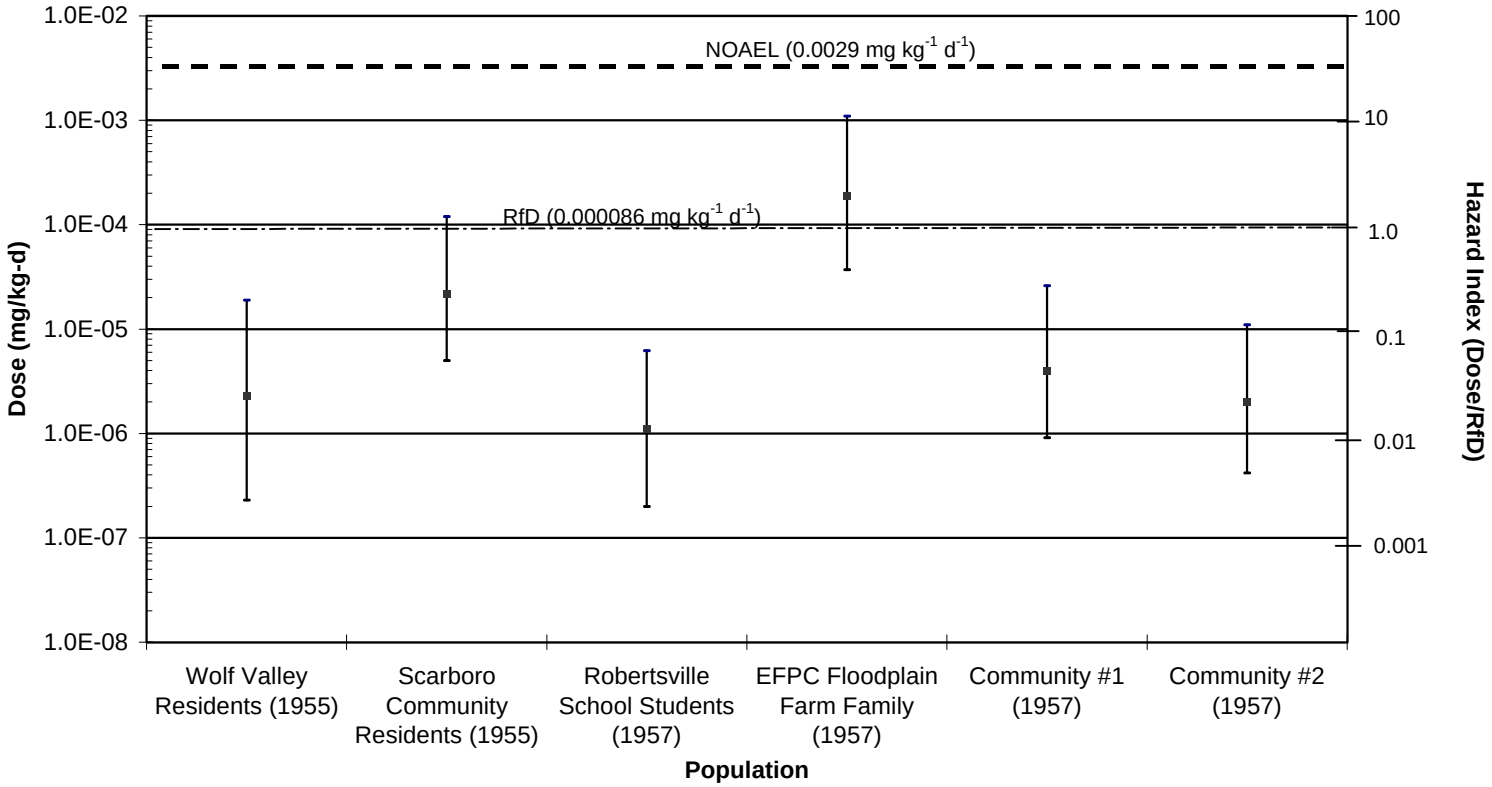


Figure 12-47: Elemental Mercury (Child exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values



- *Populations with the highest exposures***S** The highest estimated elemental mercury doses were to children who were members of the *EFPC Floodplain Farm Family* in 1957. The *upper bound* on the highest estimated annual average elemental mercury inhalation dose (0.0011 mg kg d^{-1} for the *EFPC Floodplain Farm Family* child in 1957) is about 13-times higher than the RfD derived from USEPA's reference concentration, but about 1/3 of the NOAEL. The *upper bound* estimates of inhalation doses are based on uncertain estimates of airborne mercury concentrations from transport of Y-12 airborne emissions over Pine Ridge and emission of elemental mercury from the waters of EFPC.

  Estimated doses from inhalation for the *Scarboro Community* population during 1953-1962 (when air concentrations at this location were assumed to result from both direct airborne mercury releases from Y-12 that were transported over Pine Ridge, and volatilization of mercury from EFPC) are about 15% to 40% of the inhalation doses estimated for the *EFPC Floodplain Farm Family* population during these years. During other years, estimated doses at Scarboro are about 10% of doses estimated at the EFPC Floodplain Farm Family location. The higher estimated doses at the EFPC Floodplain Farm Family location are due to its closer proximity to EFPC.

- *Likelihood of exposures above the RfD, Scarboro Community Residents***S** The estimated size of the *Scarboro Community* population was assumed to be between 800 and 1,200 individuals per year. Since estimated doses at the 50^{th} percentile for this population were below the RfD for all years, it is likely that doses to most individuals in this population were below the RfD. However, because of the relatively large size of this population, it is possible that inhalation doses to a small number of people in this population during the years of highest mercury releases from Y-12 (1953-1962) may have exceeded the RfD.

- *Likelihood of exposures above the RfD, EFPC Floodplain Farm Family members***S** The estimated size of the EFPC Floodplain Farm Family population was very small (a total of between 10 and 50 individuals were assumed in this population per year). Since estimated doses at the 50^{th} percentile to some members of this population exceeded the RfD during the years of highest mercury releases from Y-12, it is likely that doses to some individuals in this population exceeded the RfD.

Ingestion and contact with inorganic mercury in soil, sediment, water, meat, milk, and fruits/ vegetables:

(Figures 12-48 and 12-49 show how the highest estimated inorganic mercury doses for adults and children for each population compare to the RfD and the NOAEL)

- *Comparison to RfDs***S** The 95% subjective confidence interval on estimated inorganic mercury doses exceeded the USEPA RfD for inorganic mercury for at least one year for all six non-angler populations evaluated in this assessment: *Wolf Valley Residents* (child**S** 1955), the *Scarboro Community* (child**S** 1953-1962, adult**S** 1954-1959), *Robertsville School Students* (general student**S** 1955-1956, 1958; recreator**S** 1955-1958), the *EFPC Floodplain Farm Family* (child**S** 1950-1970, 1973, adult**S** 1952-1965), and the two *Oak Ridge Community* populations (Community Population #1 child**S** 1955, 1957-1958, Community Population #2 child**S** 1958).

- *Comparison to NOAELs***S** The 95% subjective confidence interval on estimated annual average inorganic mercury doses for *all* populations and *all* years were below the NOAEL for inorganic mercury. The NOAEL for inorganic mercury is based on kidney effects observed in rats fed high concentrations of water soluble mercuric chloride. The USEPA RfD is about 1,000 thousand to 3,000 times lower than reported NOAELs, because it incorporates a conservative margin of safety to account for the lack of data on the toxicity of inorganic mercury to humans. Health effects in humans exposed to inorganic mercury at doses at or below the NOAEL have not been reported.

- *Populations with the highest exposures***S** The highest estimated inorganic mercury doses were to children who were members of the *EFPC Floodplain Farm Family* in 1958. The *upper bound* on the highest estimated annual average inorganic mercury dose (0.027 mg kg d^{-1} for the *EFPC Floodplain Farm Family* child in 1958) is about 90-times higher than the USEPA RfD, but about 1/4 of the NOAEL. Doses to these individuals were estimated to be high because they were assumed to live close to EFPC on the edge of the floodplain and to be exposed through multiple pathways, including contact with contaminated soil, sediment, and water, and ingestion of "backyard" fruits/vegetables, milk, and meat. Inorganic mercury doses to *Scarboro Community Residents* during the mid- 1950s to early-1960s were also estimated to potentially exceed the RfD, because it was assumed that they occasionally recreated in EFPC (at a location only about 1 to 1 ½-mile downstream of Y-12) and consumed "backyard" fruits/vegetables.

Figure 12-48: Inorganic Mercury (Adult exposure)-
Comparison of Highest Estimated Doses to Toxicity Benchmark Values

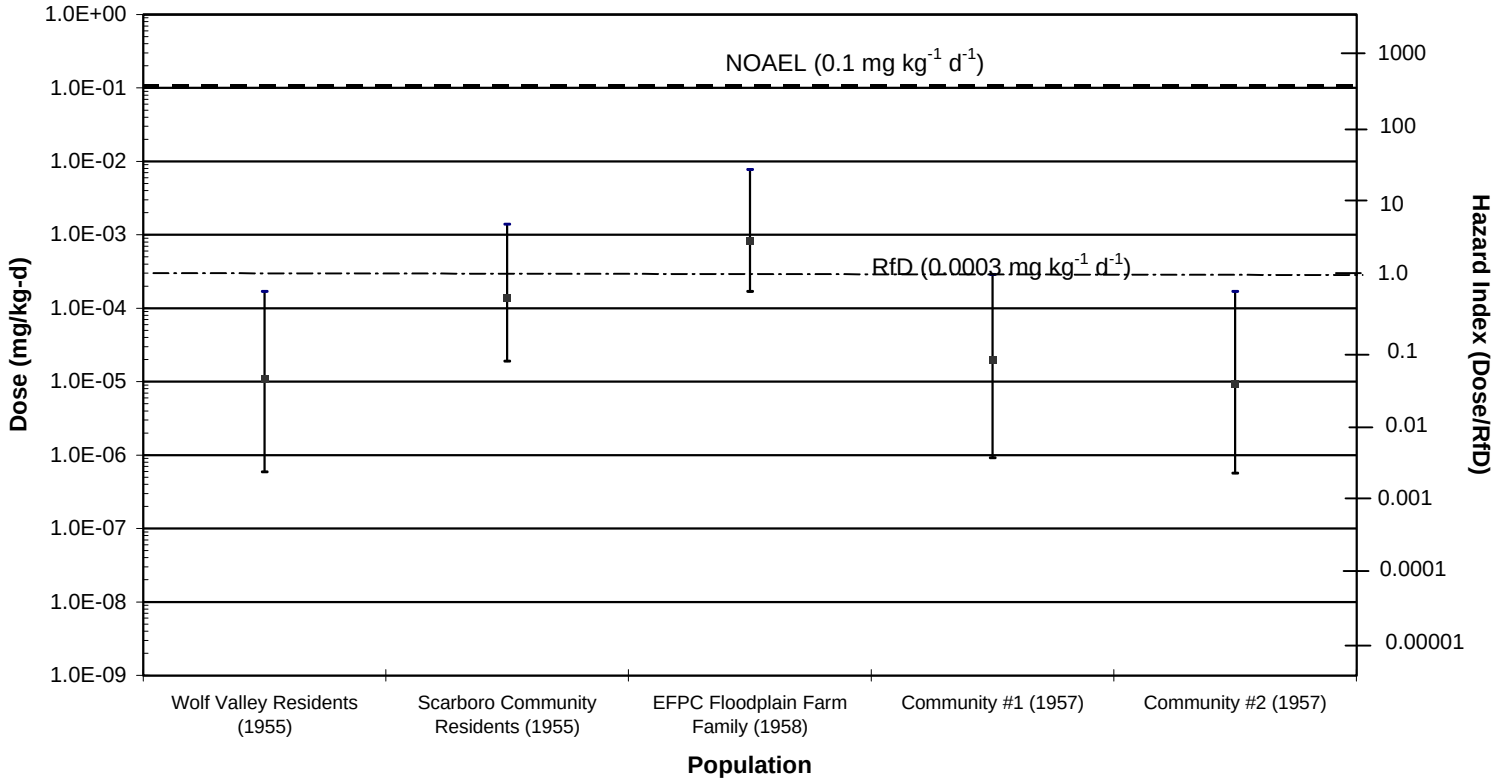
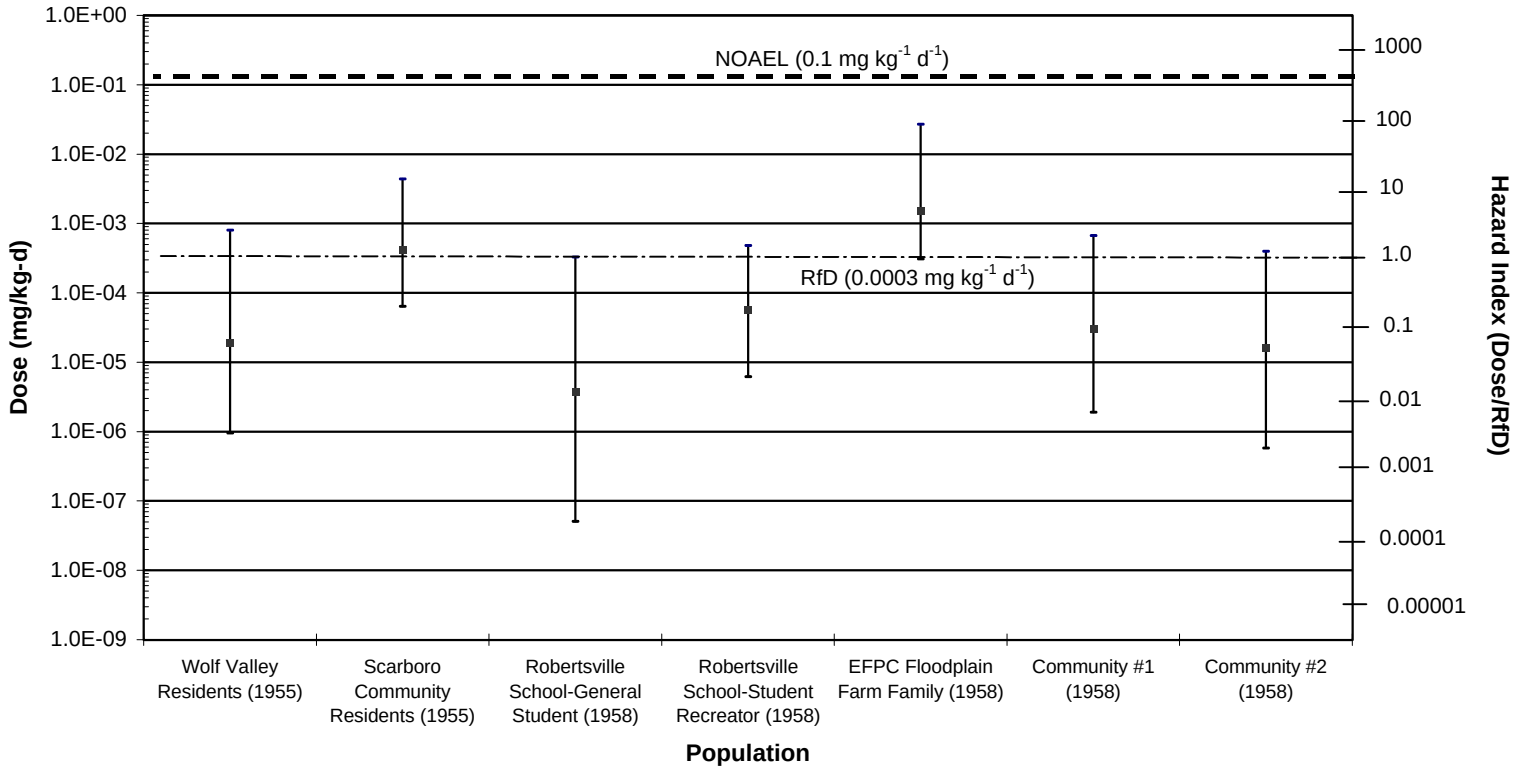


Figure 12-49: Inorganic Mercury (Child exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values



- *Important pathways***S** At five of the six locations where estimated total inorganic mercury doses exceeded the RfD, estimated doses were largely contributed by ingestion of homegrown fruits and vegetables contaminated by airborne mercury. This pathway was not evaluated for the *Robertsville School Students*; for this population, exposures were dominated by contact with contaminated surface soil and contact with contaminated water in EFPC. Contact with contaminated water in EFPC was also an important pathway for *Scarboro Community Residents* and *EFPC Floodplain Farm Family* members.

- *Likelihood of exposures above the RfD, Wolf Valley Residents***S** The estimated size of the *Wolf Valley Residents* population was small (between 30 to 100 people in a given year). For this population, the results of this assessment suggest that doses to young children *only* may have exceeded the RfD, and only if they consumed very large quantities of homegrown above-ground fruits and vegetables. Because of the small size of this population and the relatively low doses estimated for them, it is likely that the number of individuals in this population who were exposed to inorganic mercury at doses above the RfD, if any, was small.

- *Likelihood of exposures above the RfD, Scarboro Community Residents***S** The estimated size of the *Scarboro Community Residents* population was relatively large (between 800 and 1,200 individuals in a given year). Since estimated doses at the 50th percentile for this population were below the RfD for most years, it is likely that doses to most individuals in this population were below the RfD. However, because of the relatively large size of this population, it is possible that inorganic mercury doses to a moderate number of people in this population during the years of highest mercury releases from Y-12 (1953-1962) may have exceeded the RfD, particularly for those individuals who frequently recreated in EFPC or regularly consumed above-ground fruits/vegetables from backyard gardens.

- *Likelihood of exposures above the RfD, Robertsville School Students***S** The estimated size of the *Robertsville School* general student population was relatively large (between 1,500 and 2,000 students in a given year). Since estimated doses at the 50th percentile for this population were below the RfD for all years, and doses at the 97.5th percentile exceeded the RfD only during a few years in the mid-1950s, it is likely that the number of individuals in this population who were exposed to inorganic mercury at doses above the RfD was small. Behaviors most likely to have resulted in doses above the RfD were frequent contact with schoolyard soil, particularly near EFPC, and frequent contact with EFPC water and sediment.

- *Likelihood of exposures above the RfD, EFPC Floodplain Farm Family members***S** The estimated size of the *EFPC Floodplain Farm Family* population was very small (between 10 and 50 individuals in a given year). Because estimated doses at the 50th percentile for this population exceeded the RfD during the years of highest mercury releases from Y-12 (1953-1962) and because this population group was assumed to live close to EFPC, it is likely that doses to some individuals in this population exceeded the RfD. Behaviors most likely to have resulted in doses above the RfD were frequent contact with floodplain soil and EFPC water and sediment, and consumption of "backyard" fruits and vegetables.

- *Likelihood of exposures above the RfD, Community Populations***S** The estimated size of the *Community Populations* was relatively large (between 1,500 and 2,000 individuals in a given year). However, the results of this assessment suggest that for these populations, doses to young children *only* may have exceeded the RfD if they consumed very large quantities of homegrown above-ground fruits and vegetables during the years of highest mercury releases from Y-12 (mid-1950s) *and* lived closer than one-mile to the creek. Consequently, it is likely that the number of individuals in these populations who were exposed to inorganic mercury at doses above the RfD, if any, was small.

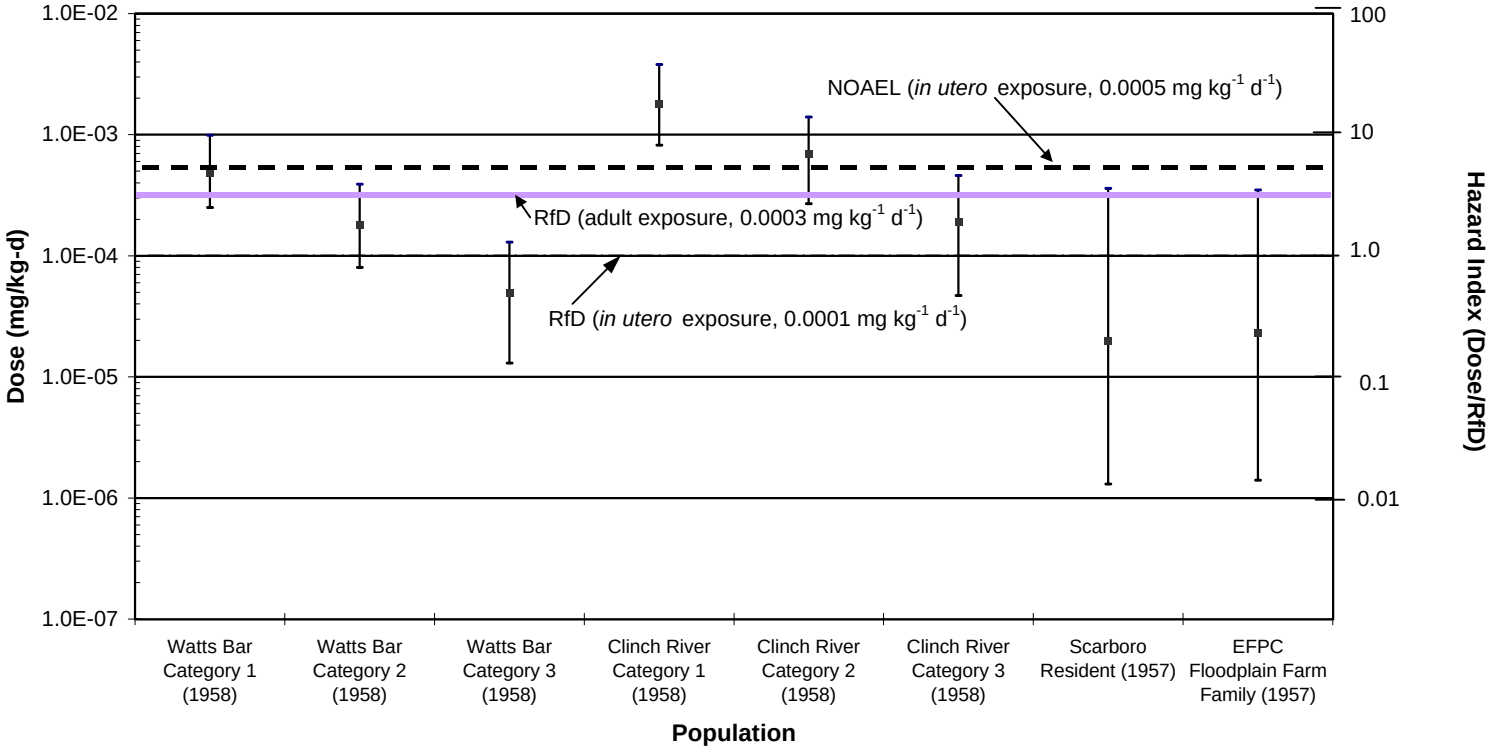
Ingestion of methylmercury in fish:

(Figure 12-50 shows how the highest estimated methylmercury doses for each population compare to the RfDs and the NOAEL).

*Consumers of Fish from Watts Bar Reservoir***S**

- *Comparison to RfDs***S** The 95% subjective confidence interval on estimated methylmercury doses from consumption of fish exceeded the USEPA RfD based on *in utero* exposures for all years for Category 1 fish consumers, 1950-1980 for Category 2 fish consumers, and 1957-1959 for Category 3 fish consumers. During the years of highest mercury releases from Y-12 (1956-1960), estimated doses for Category 1 fish consumers exceeded the RfD based on *in utero* exposures even at the *lower bound* of the distribution (the 2.5th percentile).

Figure 12-50: Methylmercury (Adult exposure)-
Comparison of Highest Estimated Doses for Each Population to Toxicity Benchmark Values



- *Comparison to NOAELs***S** The 95% subjective confidence interval on estimated methylmercury doses exceeded the NOAEL for 1956-1960 for Category 1 fish consumers. Estimated doses to Category 2 and 3 fish consumers were below the NOAEL. The NOAEL for methylmercury is based on observations of neurological effects in children who were exposed to methylmercury *in utero* when their mothers consumed methylmercury in fish during pregnancy. Health effects in humans exposed to methylmercury at doses at or below the NOAEL have not been reported.

- *Exposures to children***S** Children who ate as few as 3 to 4 meals of fish from Watts Bar Reservoir during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate 7 or more meals of fish per year from Watt Bar Reservoir during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

- *Exposures to adults***S** Adults who ate 9 or more meals of fish from Watts Bar Reservoir during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate about 20 or more meals per year during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD. Adults who were not pregnant could have consumed about three times as many fish meals per year as pregnant adult females, without being at risk of adverse health effects from methylmercury exposure, because it is believed that adults are not as sensitive to adverse health effects from methylmercury exposure as children who were exposed *in utero*.

- *Likelihood of exposures above the RfD***S** The estimated size of the recreational angler population in Watts Bar Reservoir was large (between 10,000 and 30,000 individuals in a given year). Because Watts Bar Reservoir was a productive and popular recreational fishery, it is likely that a significant number of people annually consumed a large number of fish from this system and, particularly during the mid-1950s and 1960s, were exposed to methylmercury at doses that exceeded the USEPA RfD.

*Consumers of Fish from Clinch River/ Poplar Creek***S**

- *Comparison to RfDs***S** The 95% subjective confidence interval on estimated methylmercury doses from consumption of fish exceeded the USEPA RfD based on *in utero* exposures for all years for Category 1 fish consumers, 1950-1982 for Category 2 fish consumers, and 1950-1966 for Category 3 fish consumers. Estimated doses exceeded the RfD based on *in utero* exposures even at the

lower bound of the distribution (the 2.5th percentile) for 1950-1975 for Category 1 fish consumers and 1950-1964 for Category 2 fish consumers.

- *Comparison to NOAELs***S** The 95% subjective confidence interval on estimated methylmercury doses exceeded the NOAEL for 1950-1975 for Category 1 fish consumers, 1950-1964 for Category 2 fish consumers, and 1957 for Category 3 fish consumers. The NOAEL for methylmercury is based on observations of neurological effects in children who were exposed to methylmercury *in utero* when their mothers consumed methylmercury in fish during pregnancy. Health effects in humans exposed to methylmercury at doses at or below the NOAEL have not been reported.

- *Exposures to children***S** Children who ate as few as 1 meal of fish from Clinch River/ Poplar Creek during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate about 2 or more meals of fish per year from Clinch River/Poplar Creek during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

- *Exposures to adults***S** Adults who ate 2 to 3 or more meals of fish from Clinch River/Poplar Creek during the mid- to late-1950s may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate 5 or more meals per year during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD. Adults who were not pregnant could have consumed about three times as many fish meals per year as pregnant adult females, without being at risk of adverse health effects from methymercury exposure because it is believed that adults are not as sensitive to adverse health effects from methylmercury exposure as children who were exposed *in utero*.

- *Likelihood of exposures above the RfD***S** The estimated size of the recreational angler population in Clinch River/Poplar Creek was large (between 3,000 and 10,000 individuals in a given year). Because a large number of people occasionally fished in Clinch River/Poplar Creek and many likely consumed moderate quantities of fish from this system, it is likely that a significant number of people who caught and consumed fish from this system were exposed to methylmercury at doses that exceeded the USEPA RfD, particularly if they consumed fish from this system during the mid-1950s and 1960s.

*Consumers of Fish from EFPC**S***

- *Comparison to RfDs and NOAELs**S*** The 95% subjective confidence interval on estimated methylmercury doses from consumption of EFPC fish by members of the *Scarboro Community Residents* and *EFPC Floodplain Farm Family* populations exceeded the USEPA RfD for methylmercury (based on *in utero* exposures) for all years evaluated in this assessment (1950-1990) at the 97.5th percentile. However, doses for this population did not exceed the NOAEL.

  Interviews with Oak Ridge area residents, including residents of the Scarboro Community and people who historically lived near EFPC, suggest that the maximum rate of consumption of fish from EFPC was about one fish meal per month. Consequently, Category 3 is the only category of fish consumer likely to have existed for EFPC. In this assessment, the average consumption rate of fish from EFPC for adults was assumed to be about 2.5 meals per year.

- *Exposures to children**S*** Children who ate more than 1 meal of fish per year from EFPC may have been exposed to methylmercury at doses that exceeded the USEPA RfD. If they ate 2 or more meals of fish per year from EFPC during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

- *Exposures to adults**S*** Adults who ate 2 to 3 or more meals of fish per year from EFPC may have been exposed to methylmercury at doses that exceeded the USEPA RfD based on *in utero* exposures. If they ate more than 5 meals per year during these years, it is *likely* that they were exposed to methylmercury at doses that exceeded the USEPA RfD.

Based on the results of the dose reconstruction for mercury and the comparison of estimated doses to toxicity benchmark values, the Task 2 team concluded that the following behaviors may have resulted in exposure to mercury at annual average doses above the RfDs:

Behaviors that may have resulted in exposure to mercury at annual average doses above the RfDs	
~	**Consumption of any fish** from EFPC, the Clinch River, or Poplar Creek
~	**Consumption of more than 3 or 4 meals of fish** per year from Watts Bar Reservoir
~	**Consumption of fruits or vegetables that grow above-ground from backyard gardens** in the Scarboro Community or within several hundred yards of the EFPC floodplain
~	**Playing in EFPC** more than 10-15 hours per year
~	**Living or attending school** within several hundred yards of the EFPC floodplain or in the Scarboro Community (from inhalation of airborne mercury)

The likelihood that these behaviors resulted in annual average doses above the RfDs was greatest during the period of highest mercury releases from Y-12 (that is, the mid-1950s to early-1960s).

While the results of the dose reconstruction for mercury indicate that exposures through inhalation, consumption of above-ground fruits and vegetables, contact with EFPC water and sediment, contact with EFPC floodplain soil, and consumption of fish may have resulted in annual average doses above the RfDs for mercury for some populations and some years, the results also show that annual average doses through some exposure pathways were likely insignificant, even during the years of highest mercury releases from Y-12. Based on the results of the dose reconstruction for mercury, the Task 2 team concluded that the following behaviors were *not likely* to have resulted in exposure to mercury at annual average doses above the RfDs:

Behaviors *not likely* to have resulted in exposure to mercury at annual average doses above the RfDs	
~	**Consumption of beef** from cattle that grazed in the floodplain or downwind of Y-12
~	**Consumption of fruits or vegetables from backyard gardens** located more than one mile from the EFPC floodplain (with the exception of the Scarboro Community during the 1950s and early-1960s)
~	**Living or attending school** more than 1-mile from the EFPC floodplain (from inhalation of airborne mercury; with the exception of the Scarboro Community during the 1950s and early-1960s)

This page intentionally left blank.

13.0 REFERENCES

ADP Chronology 1950-54. Author, date unknown. *Chronology of the Alloy Development Plant Development Program 1950-54*. (ChemRisk Repository No. 3265).

ADP History 1948-51. Author unknown 1956. *History of Alloy Development Project 1948-1951*. Y-F40-34/del rev. (ChemRisk Repository No. 3265).

Albers et al., 1988. J.W. Albers, L.R. Kallenbach, L.J. Fine, G.D. Langolf, R.A. Wolfe, P.D. Donofrio, A.G. Alessi, K.A. Stolp-Smith, M.B. Bromberg, and the Mercury Workers Study Group. Neurological abnormalities associated with remote occupational elemental mercury exposure. *Ann Neurol*. 24: 651-659.

Allen et al. 1994a. B.C. Allen, R.J. Kavlock, C.A. Kimmel, and E.M. Faustman. Dose-response assessment for developmental toxicity II. Comparison of generic benchmark dose estimates with no observed effect levels. *Fund. Appl. Toxicol.* 23: 487-495.

Allen et al. 1994b. B.C. Allen, R.J. Kavlock, C.A. Kimmel, and E.M. Faustman. Dose-response assessment for developmental toxicity III. Statistical models. *Fund. Appl. Toxicol.* 23: 496-509.

Alpha-5 Chronology 1953-54. Author, date unknown. *Chronology of Alpha-5 Plant 1953-54*. (ChemRisk Repository No. 3265).

Andres 1984. P. Andres. IgA–IgG disease in the intestine of Brown Norway rats ingesting mercuric chloride. *Clin. Immunol. Immunopathol.* 30: 488-494.

Ansari et al. 1973. M.S. Ansari, W.J. Miller, R.P. Gentry, M.W. Neathery, and P.E. Stake. Tissue Hg-203 distribution in young Holstein calves after single oral tracer doses in organic and inorganic forms. *J. Anim. Sci*. 36(2): 415-419.

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. May 1986. (ChemRisk Repository No. 166).

ATSDR 1994. Agency for Toxic Substances and Disease Registry. *Toxicological Profile for Mercury (Update)*. Prepared by Clement International Corporation for the U.S. Department of Health and Human Services. May 1994.

ATSDR 1997. Agency for Toxic Substances and Disease Registry. *Draft Toxicological Profile for Mercury*. Prepared by Research Triangle Institute for the U.S. Department of Health and Human Services. August 1997.

ATSDR 1998. Agency for Toxic Substances and Disease Registry. *Exposure Investigation: Serum PCB and Blood Mercury Levels in Consumers of Fish and Turtles from Watts Bar Reservoir.* U.S. Department of Health and Human Services. March 5, 1998. CERLIS No. TN1890090003.

Bache et al. 1973. C.A. Bache, W.H. Gutenmann, L.E. St. John, Jr., R.D. Sweet, H.H. Hatfield, and D.J. Lisk. Mercury and methylmercury content of agricultural crops grown on soils treated with various mercury compounds. *J. Agr. Food Chem*. 21(4): 607-613.

Baes and Orton 1979. C.F. Baes and T.H. Orton. Productivity of Agricultural Crops and Forage, Yv. Ch. 3.1 In: *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*. F.O. Hoffman and C.F. Baes, eds. ORNL/ NUREG/TM-282

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 1984.

Bailey and Lee 1991. Z. C. Bailey and R.W. Lee. *Hydrogeology and Geochemistry in Bear Creek and Union Valleys, near Oak Ridge, Tennessee.* U.S. Geological Survey Water Resources Investigations Report 90-4008. Nashville, Tennessee.

Bakir et al. 1973. F. Bakir, S.F. Damluji, L. Amin-Zaki, M. Murtadha, A. Khalidi, N.Y. Al-Rawi, S. Tikriti, H.I. Dhahir, T.W. Clarkson, J.C. Smith, and R.A. Doherty. Methylmercury poisoning in Iraq. *Science*. 181: 230-241.

Barkay et al. 1992. T. Barkay, R. Turner, E. Saouter, and J. Horn. Mercury biotransformations and their potential for remediation of mercury contamination. *Biodegradation*. 3: 147-159.

Barnes and Dourson 1988. D.G. Barnes and M. Dourson. Reference dose (RfD): Description and use in health risk assessments. *Regul. Toxicol. Pharmacol.* 8: 471-486.

Barnett and Turner 1995. M.O. Barnett and R.R. Turner. *Bioavailability of Mercury in East Fork Poplar Creek Soils*. Y-12 Internal Report, Oak Ridge National Laboratory, 1995.

Barnthouse and Deppen 1996. Consumption of fish from East Fork Poplar Creek. Memorandum to Ellen Ebert, ChemRisk. September 16.

Barton et al. 1981. S.C. Barton, N.D. Johnson, and J. Christison. Atmospheric mercury deposition in Ontario. *Proc. Annu. Meet. Air Pollut. Control Assoc*. 74, Pap. JAPCA-81-60.4.

Baumann 1952. Letter from W.H. Baumann, Health Physics, to W.K. Whitson, Y-12 Production Manager, regarding special mercury study in P-Plant, Building 9201-2. September 11, 1952. MS/ChR2-0197. (ChemRisk Repository No. 3262).

Baumann 1953a. W.H. Baumann. *Solvent Air Concentration in Stacks, 9204-4.* October 1953. Y/HG-0173. (ChemRisk Repository No. 3262).

Bernaudin et al. 1981. J.F. Bernaudin, E. Druet, P. Druet, and R. Masse. Inhalation or ingestion of organic or inorganic mercurials produces auto-immune disease in rats," *Clin. Immunol. Immunopathol.* 20: 129-135.

Beauford and Barringer 1977. W. Beauford, and A.R. Barringer. Uptake and distribution of mercury within higher plants. *Physiologia Pl.* 39:261-265.

Bishop and Neary 1974. J.N. Bishop and B.P. Neary. The form of mercury in freshwater fish. p. III-25– III-29 In: *Proc. Int. Conf. on Transport of Persistent Chemicals in Aquatic Ecosystems.* 1-3 May 1974. Natl. Res. Coun. of Canada, Ottawa, Canada.

Blaylock et al. 1983. B.G. Blaylock, M.L. Frank, C.R. Olson, and R.R. Turner. *Estimation of the concentration of mercury in bluegill (Lepomis macrochirus) in the Lower End Watts Bar Reservoir from 1946 to 1988 and potential health risks to humans*. Unpublished manuscript. (ChemRisk Repository No. 1314).

Bloom 1992. N.S. Bloom. On the chemical form of mercury in edible fish and marine invertebrate tissue. *Canadian J. Fish. Aquat. Sci.* 49: 1010.

Bogle 1972. C. Bogle. Mercury in the environment. *Med. J. Australia.* 1(3): 93-94.

Broughton 1989. J.J. Broughton. *Comprehensive Plan including 1988 Update.* City of Oak Ridge, Tennessee.

Buchet et al. 1980. J.P. Buchet, H. Roels, A. Bernard, and R. Lauwerys. Assessment of renal function of workers exposed to inorganic lead, cadmium or mercury vapor. *J. Occup. Med.* 22(11): 741-750.

Calabrese and Stanek 1992. E.J. Calabrese and E.S. Stanek. A guide to interpreting soil ingestion studies. II. Qualitative and quantitative evidence of soil ingestion. *Regul. Toxicol. Pharmacol*. 13: 278-292.

Cappon 1981. Mercury and selenium content and chemical form in vegetable crops grown on sludge-amended soil. *Arch. Environm. Contam. Toxicol.* 10: 673-689.

Case 1977. *Unclassified Version of Mercury Inventory at Y-12 Plant 1950 through 1977.* Y/AD-428. June 9, 1977. (ChemRisk Repository No. 957).

Cataldo and Wildung 1978. D.A. Cataldo and R.E. Wildung. Soil and plant factors influencing the accumulation of heavy metals by plants. *Environ. Health Perspect*. 27: 149-159.

Center 1953. Letter from C.E. Center, Vice President Union Carbide, to S.R. Sapirie, USAEC, regarding mercury for the Alloy Development Plant. March 18, 1953. Y/HG-0514. (ChemRisk Repository No. 3265).

Center 1958. Letter from C.E. Center, Vice President Union Carbide Nuclear Company, to S.R. Sapirie, Manager Oak Ridge Operations, regarding Y-12 environmental monitoring procedures. January 21, 1958. (ChemRisk Repository No. 3261 or 2657).

Chamberlain 1970. A.C. Chamberlain. Interception and retention of radioactive aerosols by vegetation. *Atmos. Environ*. 4: 57-78.

Chaney et al. 1989. R.L. Chaney, H.W. Mielke, and S.B. Sterrett. Speciation, mobility, and bioavailability of soil lead. In: *Lead in Soil: Issues and Guidelines*. B.E. Davis and B.G. Wixson, eds. Science Reviews Ltd, Northwood, England. Pp. 105-129.

ChemRisk 1993a. *Oak Ridge Health Studies Phase I Report, Vol. II - Part A - Dose Reconstruction Feasibility Study. Tasks 1 & 2: A Summary of Historical Activities on the Oak Ridge Reservation with Emphasis on Information Concerning Off-Site Emissions of Hazardous Materials*. Prepared for the Tennessee Department of Health. September 1993.

ChemRisk 1993b. *Oak Ridge Health Studies Phase I Report, Vol. 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.

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 19, 1996.

Clarkson et al. 1976. T.W. Clarkson, L. Amin-Zaki, and S.K. Al-Tikriti. An outbreak of methylmercury poisoning due to consumption of contaminated grain. *Federation Proc*. 35: 2395-2399.

Clarkson et al. 1985. T.W. Clarkson, C. Cox, D.O. Marsh, G.J. Myers, S.K. Al-Tikriti, L. Amin-Zaki, and A. R. Dabbagh. Dose-response relationships for adult and prenatal exposures to methylmercury. pp. 111-130 In: *Measurements of Risks*. G.G. Berg and H.D. Maillie, eds. Plenum, NY.

Clarkson 1990. Human health risks from methylmercury in fish. *Environ. Toxicol. Chem*. 9: 957-961.

Clewett 1953. Letter report from G.H. Clewett to L.B. Emlet regarding a review of the purpose and status of the Orex Development Program. December 7, 1953. ORNL/CF 53-12-36. (ChemRisk Repository No. 3269).

Code Words 1962. Author unknown. *Code Words*. ORO116450. October 29, 1962. (ChemRisk Repository No. 3265).

Cole et al. 1992. H.S. Cole, A.L. Hitchcock, and R. Collins. *Mercury warning– The fish you catch may be unsafe to eat: A study of mercury contamination in the United States.* Clean Water Fund, Clean Water Action, 1320 18th Street, NW, Washington, DC.

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. ORNL/ ER-155. December 1992. (ChemRisk Repository No. 1078).

Cox et al. 1989. C. Cox, T.W. Clarkson, D.O. Marsh, L. Amin-Zaki, S. Tikriti, and G.G. Myers. Dose-response analysis of infants prenatally exposed to methyl mercury: An application of a single compartment model to single-strand hair analysis. *Environ. Res.* 49: 318-332.

CRIFTC 1994. *A Fish Consumption Survey of the Umatilla, Nez Perce, Yakima, and Warm Springs Tribes of the Columbia River Basin*. Columbia River Inter-Tribal Fish Commission, Portland, OR. Technical Report 94-3.

Crump et al. 1996. Crump, K., T. Kjellstrom, A. Shipp, A. Silvers, A. Stewart. Influence of prenatal mercury exposure upon scholastic and psychological test performance determined in a statistical reanalysis. Abstract from the International Mercury Conference in Hamburg, Germany 1996.

DaMassa 1995. Memoranda from C. DaMassa to T. Mongan regarding Oak Ridge area demography. January 1, 1995.

Danielsson et al. 1993. Danielsson, B.R.G, A. Fredriksson, L. Dahlgren. Behavioural effects of prenatal metallic mercury inhalation exposure in rats. *Neurotoxicol. Teratology*. 15: 391-396.

Davidson et al. 1995. Davidson, P.W. G.J. Myers, C. Cox, et al. Longitudinal neurodevelopmental study of Seychellois children following in utero exposure to methylmercury from maternal fish ingestion: outcomes at 19 and 20 months. *Neurotoxicol.* 16(4): 677-688.

Davis et al. 1996. A. Davis, N.S. Bloom, and S.S. Que Hee. The environmental geochemistry and bioaccessibility of mercury in soil and sediments: A review. Submitted to *Risk Analysis.*

De Temmerman et al. 1986. L. de Temmerman, R. Vandeputte, and M. Guns. Biological monitoring and accumulation of airborne mercury in vegetables. *Environ. Pollut. Ser. A* 41: 139-151.

Dill 1967. M.S. Dill. *Determination of Submicrogram Quantities of Mercury in Water and Lithium Hydroxide Solutions*. March 28, 1967. Y-1572. (ChemRisk Repository No. 3261).

Dourson et al. 1996. Evolution of science-based uncertainty factors in noncancer risk assessment. *Regul. Toxicol. Pharmacol.* 24: 108-120.

Dragun and Chiasson 1991. J.D. Dragun and A. Chiasson. *Elements in North American Soils.* Hazardous Materials Control Resources Institute. Greenbelt, MD.

Dreicer et al. 1984. M. Dreicer, T.E. Hakonson, G.C. White, and F.W. Whicker. Rainsplash as a mechanism for soil contamination of plant surfaces. *Health Phys*. 46(1): 177-187.

Dreicer et al. 1990. M. Dreicer, A. Bouville, B.W. Wachholz. Pasture practices, milk distribution, and consumption in the continental U.S. in the 1950s. *Health Phys*. 59(5): 627-636.

Druet et al. 1978. P.E. Druet, F. Druet, F. Potdevin, and C. Sapin. Immune type glomerulonephritis induced by $HgCl_2$ in the Brown Norway rat," *Ann. Immunol*. 129C: 777-792.

Eastern Research Group, Inc. 1988. *U.S. Environmental Protection Agency Peer Review Workshop on Mercury Issues, Cincinnati, Ohio, October 26-27, 1987. Summary Report*. February 5 1988.

Ebert 1996. E.S. Ebert. Fish consumption distributions for populations of interest for the dose reconstruction. Memo to T.E. Widner, Mclaren-Hart. April 26.

Edwards 1972. Letter from R.S. Edwards, Industrial Hygiene, to H. Postma, Thermonuclear Division UCCND, regarding mercury survey in Building 9201-2. June 8, 1972. Y/HG-0241. (ChemRisk Repository No. 3262).

Ehrenberg et al. 1991. R.L. Ehrenberg, R.L. Vogt, A.B. Smith, J. Brondum, W. S. Brightwell, P.J. Hudson, K.P. McManus, W.H. Hannon, F.C. Phipps. Effects of elemental mercury exposure at a thermometer plant. *Am. J. Ind. Med.* 19: 495-507.

Elwood 1977. J.W. Elwood. *Mercury Contamination in Poplar Creek and Clinch River*. ORNL/CF-77/320. June 6, 1977. (ChemRisk Repository No. 3052).

Elwood 1984. J.W. Elwood. *Mercury Contamination in Poplar Creek and the Clinch River*. Environmental Sciences Division. ORNL/TM-8893. February 17, 1984. (ChemRisk Repository No. 172).

Faustman et al. 1994. E.M. Faustman, B.C. Allen, R.J. Kavlock, and C.A. Kimmel. Dose-response assessment for developmental toxicity I. Characterization of database and determination of no observed effect levels. *Fund. Appl. Toxicol.* 23: 478-486.

Fawer et al. 1983. R.F. Fawer, U. DeRibaupierre, M.P. Guillemin, M. Berode, and M. Lobe. Measurement of hand tremor inducted by industrial exposure to metallic mercury. *J. Ind. Med.* 40: 204-208.

Fee and Sanders 1982. Letter from G.G. Fee, Oak Ridge Y-12 Plant Manager, to H.D. Hickman, Department of Energy Oak Ridge Operations, regarding mercury water analyses from the outfall of New Hope Pond at the headwaters of East Fork Poplar Creek from 1954 to 1982. Includes tables prepared by M. Sanders. December 10, 1982. Y/TS-1610. (ChemRisk Repository No. 3259).

Fimreite and Reynolds 1973. N. Fimreite and L.M. Reynolds. Mercury contamination of fish in northwestern Ontario. *J. Wild. Manage.* 37(1): 62-68.

Finley et al. 1994. B. Finley, D. Proctor, P. Scott, N. Harrington, D. Paustenbach, and P. Price. Recommended distributions for exposure factors frequently used in health risk assessment. *Risk Anal.* 14(4)533-553.

Fitzgerald 1986. W.F. Fitzgerald. *The Role of Air-Sea Exchange in Geochemical Cycling.* NATO Advanced Science Institutes Series, P. Buat-Menard, ed. Reidel, Dordrecht. pp. 363-408.

Fitzgerald 1989. W.F. Fitzgerald. Atmospheric and ocean cycling of mercury. Ch. 57 In: *Chemical Oceanography*. Academic Press, New York.

Fitzgerald and Clarkson 1991. W.F. Fitzgerald and T.W. Clarkson. Mercury and monomethylmercury–Present and future concerns. *Environ. Health Perspect.* 96: 159-166.

Ford 1983. Memorandum from R.T. Ford, Industrial Hygiene Department, to G.E. Isham, Building 9204-2E, regarding air sampling of the mercury-thallium operation in Building 9204-2. May 18, 1983. (ChemRisk Repository No. 3262).

Fortmann et al. 1977. L.C. Fortmann, D.D. Gay, and K.O. Wirtz. Ethylmercury. Formation in plant tissue and relation to methylmercury formation. *Trace Subst. Environ. Hlth.* 11: 117.

Fraser 1995. R.J. Fraser. Interview with Mr. Raymond J. Fraser, Y-12 Classification Office, on January 27, 1995. (ChemRisk Repository No. 3270).

Fredriksson et al. 1992. A. Fredriksson, L. Dahlgren, B. Danielsson, P. Eriksson, L. Dencker, and T. Archer. Behavioural effects of neonatal metallic mercury exposure in rats. *Toxicology* 74:151-160.

Fredriksson et al. 1996. A. Fredriksson, L. Dencker, T. Archer, et al. Prenatal exposure to metallic mercury vapour and methylmercury produce interactive behavioural changes in adult rats. *Neurotoxicol. Teratol.* 18(2): 129-134.

Gerstner and Huff 1977. H.B. Gerstner and J.E. Huff. Clinical toxicology of mercury. *J. toxicol. Environ. Health* 2:491-526.

Gifford 1995. F. Gifford. Personal communication with T. Mongan on January 19, 1995.

Gilmour et al. 1992. C.C. Gilmour, E.A. Henry, and R. Mitchell. Sulfate stimulation of mercury methylation in freshwater ecosystems. *Environ. Sci. Technol.* 26(11): 2281-2286.

Gist 1986. C.S. Gist. *Well Water Sampling by Oak Ridge Associated Universities*. K-25 Compliance and Environmental Policy Document Center Report No. 801263. (ChemRisk Repository No. 507).

Gist 1987. C.S. Gist. Soil Contaminant Uptake by Plants in the Terrestrial Food Chain in the Floodplain of East Fork Poplar Creek. Oak Ridge Associated Universities.

Goyer 1996. R.A. Goyer. Toxic effects of metals. Ch. 23 In: *Casarett and Doull's Toxicology, The Basic Science of Poisons*. C.D. Klaassen, ed. 5th Ed. McGraw-Hill, New York.

Grandjean et al. 1992. Grandjean, P., P. Weihe, P.J. Jorgensen, T. Clarkson, E. Cernichiari, T. Videro. Impact of maternal seafood diet on fetal exposure to mercury, selenium, and lead. *Arch. Environ. Health* 47(3): 185-195.

Guettner 1971. Memorandum from P.D. Guettner to D.W. Smith, Union Carbide Corporation Nuclear Division, regarding 81-10 area core samples. August 27 1971. Cited in UCCND (1983a).

HAI 1994. History Associates, Inc. *Mercury Task Force Files: A Guide to Record Series of the Department of Energy and its Contractors*. Draft. October 12, 1994. (ChemRisk Repository No. 3272).

Hargis, H.L. 1968. *Development of Improved Fishing Methods for Use in Southeastern and South-Central Reservoirs: Review of the Current Status of the Commercial Fishery in Tennessee.* Financed with Federal Aid Funds Under Provisions of the Commercial Fisheries Research and Development Act of 1964 (Public Law 88-309). Job Completion Report, 4-5-R-2. March.

Harris and Snodgrass, n.d. A mechanistic model to examine mercury in aquatic systems (II): Mercury concentrations, fluxes, and distribution in a generic Canadian shield lake. Ontario Hydro and McMaster University, Ontario, Canada. Received May 1997.

Healy 1968. W.B. Healy. Ingestion of soil by dairy cows. *N.Z. J. Agric. Res.* 11: 487-499.

Henke et al. 1993. K.R. Henke, V. Kuhnel, D.J. Stepan, R.H. Fraley, C.M. Robinson, D.S. Charlton, H.M. Gust, and N.S. Bloom. Topical Report: *Critical Review of Mercury Contamination Issues Relevant to Manometers at Natural Gas Industry Sites.* Gas Research Institute, Chicago, IL. GRI-93/0117.

Henry 1955. H.F. Henry. Memo from H.F. Henry to W.L. Richardson re: Mercury in Sanitary Water. Safety, Fire, and Radiation Control Division, Carbide and Carbon Chemicals Company, Oak Ridge, TN. September 7.

Hibbitts 1984. H.W. Hibbitts. *Transmittal of Environmental Sampling Data for Mercury*. February – December 1984. (ChemRisk Repository No. 536).

Hibbitts 1986. H.W. Hibbitts. Transmittal of Environmental Sampling Data for Mercury. 1986.

Hibbs 1966. Letter from R.F. Hibbs, Y-12 Superintendent, to C.A. Keller, USAEC, regarding stripping of Building 9201-5. April 26, 1966. Y/HG-0274. (ChemRisk Repository No. 3265).

Hildebrand et al. 1980. S.G. Hildebrand, S.E. Lindberg, R.R. Turner, J.W. Huckabee, R.H. Strand, J.R. Lund, A.W. Andren. *Biogeochemistry of mercury in a river-reservoir system: Impact of an inactive chloralkali plant on the Holston River- Cherokee Reservoir, Virginia and Tennessee*. Oak Ridge National Laboratory, Environmental Science Division. ORNL/TM-6141.

Hill 1955. Memos from G.S. Hill to A.F. Becher re: Mercury Contamination**S** Poplar Creek and Clinch River. Safety and Health Physics Division, Carbide and Carbon Chemicals Company, Oak Ridge, TN. November 2 and November 8.

Hoffman and Baes 1979. F.O. Hoffman and C.F. Baes. *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*. Oak Ridge National Laboratory, prepared for the U.S. Nuclear Regulatory Commission. ORNL/NUREG/TM-282.

Horowitz 1991. A.J. Horowitz. *A Primer on Sediment-Trace Element Chemistry*. 2nd edition. Lewis Publishers, Chelsea, MI.

Horowitz and Finley 1994. S.B. Horowitz and B.L. Finley. Setting health-protective soil concentrations for dermal contact allergens: A proposed methodology. *Regul. Toxicol. Pharmacol*. 19: 31-47.

HSDB 1998. Hazardous Substances Data Bank. Files for mercury (CAS# 7439-97-6) and methylmercury (CAS# 22967-92-6). National Library of Medicine, Washington DC.

Huckabee et al. 1975. J.W. Huckabee, R.A. Goldstein, S.A. Janzen, and S.E. Woock. Methylmercury in a freshwater foodchain. pp. 199-216. In: *Proc. of Int. Conf. on Heavy Metals in the Environment.* Toronto, Ontario, Canada. Oct. 27-31, 1975.

Huckabee et al. 1979. Accumulation of mercury in freshwater biota. Ch. 12 In: *The Biogeochemistry of Mercury in the Environment.* Elsevier/North-Holland Biomedical Press.

IAEA 1992. *Modelling of Resuspension, Seasonality, and Losses during Food Processing*. First report of the VAMP Terrestrial Working Group. IAEA-TECDOC-647. Vienna, Austria. May 1992.

IAEA 1994. *Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments*. Technical Report Series No. 364. Vienna, Austria. 1994.

IRIS 1998. Integrated Risk Information System. Toxicity Profiles for Mercury (Inorganic), Elemental, and Methylmercury. National Library of Medicine Database.

Israeli and Nelson 1992. M. Israeli and C.B. Nelson. Distribution and expected time of residence for U.S. households. *Risk Anal*. 12(1): 65-72.

Jennings 1965. Memorandum from D.A. Jennings, Union Carbide, to J.W. Ebert, Union Carbide, regarding the stripping of Building 9201-5. June 4, 1965. Y/HG-0246. (ChemRisk Repository No. 3265).

Kabata-Pendias and Pendias 1984. A. Kabata-Pendias and H. Pendias. *Trace Elements in Soils and Plants*. CRC Press, Inc. Boca Raton, Florida.

Keyes 1994. J.E. Keyes. *Oak Ridge Y-12 Plant Large-Scale Review Plan*. Y/SA-855. July 1994. (ChemRisk Repository No. 3272).

Kite 1957. Letter from H.T. Kite to N.H. Bethea regarding solvent inventory in Building 9201-2. December 13, 1957. Y/HG-0314. (ChemRisk Repository No. 3265).

Kite 1958. Letter from H.T Kite to N.H. Bethea regarding stripping of the pilot plant facility in Building 9201-2. June 27, 1958. Y/HG-0447. (ChemRisk Repository No. 3265).

Kitson 1945. Memorandum from R.E. Kitson to B.F. Butler regarding a summary of the work done on the mercury cathode. July 26, 1945. ORNL/CF 45-7-476. (ChemRisk Repository No. 3269).
Kjellstrom et al. 1986. T. Kjellstrom, P. Kennedy, S. Wallis, and C. Mantell. *Physical and Mental Development of Children with Prenatal Exposure to Mercury from Fish. Stage 1: Preliminary Tests at Age 4.* National Swedish Environmental Protection Board, Report 3080.

Kjellstrom et al. 1989. T. Kjellstrom, P. Kennedy, and S. Wallis. *Physical and Mental Development of Children with Prenatal Exposure to Mercury from Fish Stage 2: Interviews and Psychological Tests at Age 6.* National Swedish Environmental Protection Board, Report 3642.

Koranda 1965. J.J. Koranda. *Agricultural Factors Affecting the Daily Intake of Fresh Fallout by Dairy Cows*. Lawrence Radiation Laboratory, Livermore, CA. March 19.

Kozuchowski and Johnson 1978. J. Kozuchowski and D.L. Johnson. Gaseous emissions of mercury from an aquatic vascular plant. *Nature, Lond*. 274: 468-469.

Kwasnoski and Whitson 1955 1956 1957. T. Kwasnoski and T.C. Whitson. *Water Sampling Program for Mercury, September 1955 - September 1957.* KLI-3654, KLI-3693, KLI-3705, KLI-3621, KLI-4236. (ChemRisk Repository No. 783).

LaFrance 1957. L.J. LaFrance. *Preliminary Report on Personnel Exposure to Mercury in the Colex Plants*. May 28, 1957. Y-D1-1. (ChemRisk Repository No. 3262).

LaFrance 1955-60. L.J. LaFrance. Monthly Solvent Air Analysis Reports, April 1955 to February 1960. MS/ChR2-0242/del rev. (ChemRisk Repository No. 3262).

LaGrone 1983. J. LaGrone. Includes a written statement and the transcript of the actual testimony of Joseph LaGrone, Manager, U.S. DOE Oak Ridge Operations, included in Lloyd and Gore 1983. July 11, 1983. (ChemRisk Repository No. 3278).

Landis 1976. Memorandun from S.D. Landis, Industrial Hygiene, to R. Edwards, Building 9201-2, regarding mercury survey in basement of Building 9201-2. March 22, 1976. Y/HG-0241. (ChemRisk Repository No. 3262).

Langworth et al. 1992. S. Langworth, O. Almkvist, E. Söderman, and B-O. Wikström. Effects of occupational exposure to mercury vapour on the central nervous system. *Br. J. Ind. Med.* 49: 545-555.

Lauwerys et al. 1983. R. Lauwerys, A. Bernard, H. Roels, J.P. Buchet, J.P. Gennart, P. Mahieu, and J.M. Foidart. Anti-laminin antibodies in workers exposed to mercury vapour. *Toxicol. Lett.* 17: 113-116.

Lauwerys et al. 1985. R. Lauwerys, H. Roels, P. Genet, G. Toussaint, A. Bouckaert, and S.De Cooman. Fertility of male workers exposed to mercury vapor or to manganese dust: A questionnaire study. *Am. J. Ind. Med.* 7: 171-176.

Layton, 1993. D.W. Layton. Metabolically consistent breathing rates for use in dose assessment. *Health Phys*. 64(1): 23-26.

Levine et al. 1982. S.P. Levine, G.D. Cavender, G.D. Langolf, and J.W. Albers. Elemental mercury exposure: peripheral neurotoxicity. *Br. J. Ind. Med.* 39: 136-139.

Liang et al. 1993. Y-X. Liang, R-K. Sun, Y. Sun, Z-Q. Chen, and L-H. Li. Psychological effects of low exposure to mercury vapor: Application of a computer-administered neurobehavioral evaluation system. *Environ. Res.* 60: 320-327.

Liebert et al. 1991. C.A. Liebert, T. Barkay, and R.R. Turner. Acclimation of aquatic microbial communities to Hg(II) and CH_3Hg^+ in polluted freshwater ponds. *Microb. Ecol*. 21: 139-149.

Lindberg et al. 1979. S.E. Lindberg, D.R. Jackson, J.W. Huckabee, S.A. Janzen, M.J. Kevin, and J.R. Lund. Atmospheric emission and plant uptake of mercury from agricultural soils near the Almaden mercury mine. *J. Environ. Qual*. 8(4): 572-578.

Lindberg et al. 1991. S.E. Lindberg, R.R. Turner, T.P. Meyers, G.E. Taylor, Jr., and W.H. Schroeder. Atmospheric concentrations and deposition of mercury to a decidious forest at Walker Branch watershed, Tennessee, USA. *Water, Air, Soil Pollut.* 56:577-594.

Lindberg et al. 1992. S.E. Lindberg, T.P. Meyers, G.E. Taylor, R.R. Turner, and W.H. Schroeder. Atmospheric/ surface exchange of mercury in a forest: Results of modeling and gradient approaches. *J. Geophys. Res*. 97: 2519-2528.

Lindberg et al. 1994. S.E. Lindberg, J.G. Owens, and W.J. Stratton. Application of throughfall methods to estimate dry deposition of mercury. Ch. II.8 In: *Mercury Pollution: Integration and Synthesis*. C.J. Watras and J.W. Huckabee, eds. Lewis Publishers, Ann Arbor.

Lindberg et al. 1995. S.E. Lindberg, K-H. Kim, T.P. Meyers, and J.G. Owens. A micrometeorological gradient approach for quantifying air/surface exchange of mercury vapor: Tests over contaminated soils. *Environ. Sci. Technol*. 29: 126-135.

Lindberg et al. 1996. S.E. Lindberg, P.J. Hanson, T.P. Meyers, and K-H. Kim. Air/ surface exchange of mercury over forests: A reassessment of continental biogenic mercury emissions. Submitted to *Atmos. Environ*. (9/6/1996).

Lindqvist et al. 1991. O. Lindqvist, K. Jogansson, M. Aastrup, A. Andersson, L. Bringmark. G. Hovsenius, L. Hadanson, A. Iverfeldt, M. Meili, and B. Timm. Mercury in the Swedish environment, recent research on causes, consequences, and corrective methods. *Water Air Soil Pollut.* 55: 1-253.

Little 1956. Memorandum from J.C. Little to L.E. Burkhart regarding solvent losses through ventilation exhaust systems- Building 9201-5. March 14, 1956. Y/HG-0281/del rev. (ChemRisk Repository No. 3263).

Little and Miller 1979. C.A. Miller and C.W. Miller. *The Uncertainty Associated with Selected Environmental Transport Models.* ORNL-5528.

Lloyd and Gore 1983. M. Lloyd and A. Gore, Jr. Transcript of the July 11, 1983 Congressional Hearing on the Impact of Mercury Releases at the Oak Ridge Complex. (ChemRisk Repository No. 3278).

LMES 1995. Lockheed Martin Energy Systems. *Hazard Classification for Building K-1303.* September 1995. ER033638. (ChemRisk Repository No. 3268).

Loar et al. 1981a. J.M. Loar, F.A. Burkhart, G.F. Cada, J.W. Huckabee, J.T. Kitchings, K.D. Kumar, A.M. Sasson, J.A. Soloman, and J.D. Story. *Ecological Studies in the Biotic Communities in the Vicinity of the Oak Ridge Gaseous Diffusion Plant*. Environmental Sciences Division. ORNL/TM-6714. October 1981. (ChemRisk Repository No. 250).

Loar et al. 1981b. 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 the White Oak Creek Watershed and the Clinch River below Melton Hill Dam*. ORNL/TM-7509/V2. October 1981. (ChemRisk Repository No. 154).

Loar et al. 1989. J.M. Loar, S.M. Adams, L.J. Allison, J.M. Giddings, J.F. McCarthy, G.R. Southworth, J.G. Smith, and A.J. Stewart. *The Oak Ridge Y-12 Plant Biological Monitoring and Abatement Program for East Fork Poplar Creek. Environmental Sciences Division*. ORNL/TM-10265. October 1989. (ChemRisk Repository No. 81).

Loar et al. 1992. J.M. Loar, S.M. Adams, L.A. Kszos, M.G. Ryon, J.G. Smith, G.R. Southworth, and A.J. Stewart. *Oak Ridge Gaseous Diffusion Plant Biological Monitoring and Abatement Program for Mitchell Branch*. Environmental Sciences Division. Report ORNL/TM-11965. January 1992. (ChemRisk Repository No. 224).

Loar, J.M. (editor) 1992a. *First Annual Report on the ORNL Biological Monitoring Program at Oak Ridge National Laboratory*. ORNL/TM-10399. (ChemRisk Repository No. 1752).

Loar, J.M. (editor) 1992b. *Second Annual Report on the ORNL Biological Monitoring and Abatement Program*. ORNL/TM-10804. (ChemRisk Repository No. 1748).

Loar, J.M. (editor) 1994a. *Third Annual Report on the ORNL Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River*. ORNL/TM-11358. (ChemRisk Repository No. 1749).

Loar, J.M. (editor) 1994b. *Fourth Annual Report on the ORNL Biological Monitoring and Abatement Program for White Oak Creek Watershed and the Clinch River*. ORNL/TM-11544. (ChemRisk Repository No. 1751).

Lodenius 1990. M. Lodenius. Sorption of mercury in soils. Ch. 12 In: *Encyclopedia of Environmental Control Technology. Volume 4: Hazardous Waste Containment and Treatment.* P. N. Cheremisinoff, ed. Gulf Publishing Co., Houston.

Lodenius et al. 1994. M. Lodenius. Mercury in terrestrial ecosystems: A review. Ch. III.6 In: *Mercury Pollution, Integration and Synthesis*. C. J. Watras and J.W. Huckabee, eds. Lewis Pub., Ann Arbor.

Log sheets 1957-62. Solvent Recovery Facility Log Sheets, May 1957 through October 1962. Includes Cumulative Summary of Solvent Recovery Facility April 1957 through March 1959. Y/HG-0005 and Y/HG-0023. (ChemRisk Repository No. 3266).

Marsh et al. 1980. D.O. Marsh, G.J. Myers, T.W. Clarkson, L. Amin-Zaki, S. Al-Tikriti, M.A. Majeed. Fetal methylmercury poisoning: Clinical and toxicological data on 29 cases. *Ann. Neurol.* 7:348-353.

Marsh et al. 1981. D.O. Marsh, G.J. Myers, T.W. Clarkson, L. Amin-Zaki, S. Tikriti, M.A. Majeed, and A.R. Dabbagh. Dose-response relationship for human fetal exposure to methylmercury. *Clin. Toxicol.* 18(11): 1311-1318.

Marsh et al. 1987. D.O. Marsh, T.W. Clarkson, C.C. Cox, G.J. Myers, L. Amin-Zaki, S. Al-Tikriti. Fetal methylmercury poisioning: Relationship between concentration in single strands of maternal hair and child effects. *Arch. Neurol.* 1017-1022.

Marsh et al. 1995. Marsh, D.O., T.W. Clarkson, G.J. Myers, P.W. Davidson, C. Cox, E. Cernichiari, M.A. Tanner, W. Lednar, C. Shamlaye, O. Choisy, C. Hoareau, M. Berlin. The Seychelles study of fetal methylmercury exposure and child development: Introduction. Methylmercury and Human Health- Special Issue. *Neurotoxicology*16(4): 583-596.

Marshall 1983. E. Marshall. The "lost" mercury at Oak Ridge. *Science*. 221: 131-132.

Mason et al. 1994. R.P. Mason. Elemental mercury cycling within the mixed layer of the equatorial Pacific Ocean. Ch. I.7 In: *Mercury Pollution, Integration and Synthesis*. C. J. Watras and J.W. Huckabee, eds. Lewis Publishers, Ann Arbor.

Matilda et al. 1971. Y. Matilda. Toxicity of mercury compounds to aquatic organisms and accumulation of the compounds by the organisms. *Bull. Freshw. Fish. Res. Lab*. 21: 197-227.

Mayland et al. 1977. H.F. Mayland, G.E. Shewmaker, and R.C. Bull. Soil ingestion by cattle grazing crested wheatgrass. *J. Range Mgmt*. 30(4): 264-265.

McBryde and Williams 1957. W.T. McBryde and F. Williams. *A Rapid Determination of Micro Quantities of Mercury in Urine and Water Using the Mercurometer*. September 13 1957. Y-1178. (ChemRisk Repository No. 3261).

McCauley 1995. L.L. McCauley. Interview with Mr. Lowell L. McCauley 1983 Mercury Task Force, on January 27 1995. (ChemRisk Repository No. 3270).

McKeown-Eyssen et al. 1983. McKeown-Eyssen, G., J. Ruedy, A. Neims. Methyl mercury exposure in northern Quebec II. Neurologic findings in children. *Amer. J. Epid.* 118(4): 470-479.

McKim et al. 1976. J.M. McKim. Long term effects of methylmercuric chloride on three generations of brook trout: toxicity, accumulation, distribution, and elimination. *J. Fish. Res. Board Can*. 33: 2726-2739.

McKone 1988. T.E. McKone. *Conventional Weapons Demilitarization: A Health and Environmental Effects Data Base Assessment. Methods of Estimating Multi-Pathway Exposure to Environmental Contaminants*. Final Report, Phase II. AD UCRL-21064. Lawrence Livermore National Laboratory, Livermore, CA.

McKone 1994. T.E. McKone. Uncertainty and variability in human exposures to soil contaminants through home-grown food: A Monte Carlo assessment. *Risk Anal*. 14(4): 449-463.

McMurray and Redmond 1958. C.S. McMurray and J.W. Redmond. *Portable Mercury Vapor Detector*. January 7, 1958. Y-1188. (ChemRisk Repository No. 3261).

McRee et al. 1965. P.C. McRee, C.M. West, J.D. McLendon. *Y-12 Radiation Safety Manual*. March 24, 1965. Y-1401. (ChemRisk Repository No. 3261).

Miller 1979. C.W. Miller. The Interception Fraction. Ch. 3.2 In: *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*. F.O. Hoffman and C.F. Baes, eds. ORNL/ NUREG/TM-282.

Miller and Hively 1987. C.W. Miller and L.M. Hively. A review of validation studies for the Gaussian Plume Atmospheric Dispersion Model. *Nucl. Safety*. 28(4): 522-532.

Miller and Hoffman 1979. Miller, C.W. and F.O. Hoffman. "The Environmental Loss Constant for Radionuclides Deposited on the Surfaces of Vegetation, δ_w." In: *A Statistical Analysis of Selected Parameters for Predicting Food Chain Transport and Internal Dose of Radionuclides*. F.O Hoffman and C.F. Baes, ed. U.S. Nuclear Regulatory Commission. Office of Standards and Development. NUREG/CR-1004. pp. 43-50.

Mishinova et al. 1980. V.N. Mishonova, P.A. Stepanova, and U.U. Zarudin. Characteristics of the course of pregnancy and labor in women coming in contact with low concentrations of metallic mercury vapors in manufacturing work places. *Gig Tr Prof Zabol*. Issue 2:21-23.

MMES 1984. Martin Marietta Energy Systems. *Environmental Monitoring Report. United States Department of Energy, Oak Ridge Facilities. Calendar Year 1983.* Y/UB-19. (ChemRisk Repository No. 954).

MMES 1985. Martin Marietta Energy Systems. *Environmental Monitoring Report. United States Department of Energy, Oak Ridge Facilities. Calendar Year 1984.* ORNL-6209. (ChemRisk Repository No. 955).

MMES 1986. Martin Marietta Energy Systems. *Environmental Surveillance of the Oak Ridge Reservation and Surrounding Environs during 1985*. ORNL-6271. (ChemRisk Repository No. 199).

MMES 1987. Martin Marietta Energy Systems. *RCRA Facility Investigation Plan K-1420 Mercury Room ORGDP*. November 1987. ER005565. (ChemRisk Repository No. 3268).

MMES 1988. Martin Marietta Energy Systems. *Environmental Surveillance of the U.S. Department of Energy Oak Ridge Reservation and Surrounding Environs during 1987. Volume 1: Narrative, Summary, and Conclusions*. ES/ESH-4/V1. (ChemRisk Repository No. 370).

MMES 1989. Martin Marietta Energy Systems. *Oak Ridge Reservation Environmental Report for 1988. Volume 1: Narrative, Summary, and Conclusions.* ES/ESH-8/V1. (ChemRisk Repository No. 254).

MMES 1990. Martin Marietta Energy Systems. *Oak Ridge Reservation Environmental Report for 1989. Volume 2: Data Presentation*. ES/ESH-13/V2. (ChemRisk Repository No. 227).

MMES 1991. Martin Marietta Energy Systems. *Oak Ridge Reservation Environmental Report for 1990. Volume 1: Narrative, Summary, and Conclusions.* ES/ESH-18/V1. (ChemRisk Repository No. 582).

Morehead 1957. J.F. Morehead. *Sludge Burner Stack Loss of Solvent*. June 18, 1957. Y/HG-0169/1. (ChemRisk Repository No. 3266).

Mosbaek et al. 1988. H. Mosbaek, J.C. Tjell, and T. Sevel. Plant uptake of airborne mercury in background areas. *Chemosphere*. 17(6): 1227-1236.

Mullen et al. 1975. A.L. Mullen, R.E. Stanley, S.R. Lloyd, and A.A. Moghissi. Absorption, distribution, and milk secretion of radionuclides by the dairy cow. IV. Inorganic radiomercury. *Health Phys*. 28: 685-691.

Murray 1956. Letter from J.P. Murray, Y-12 Plant Superintendent, to R.C. Armstrong, USAEC Director of Production, regarding the Y-12 steam plant, Building 9401-3. January 27 1956. (ChemRisk Repository No. 3267).

Myers et al. 1995. Myers, G.J., P.W. Davidson, C. Cox, C. Shamlaye, M.A. Tanner, O. Choisy, J. Sloan-Reeves, D.O. Marsh, E. Cernichiari, A. Choi, M. Berlin, T.W. Clarkson. Neurodevelopmental outcomes of Seychellois children sixty-six months after in utero exposure to methylmercury from a maternal fish diet: Pilot study. Methylmercury and Human Health- Special Issue. *Neurotoxicology*16(4): 639-652.

Napier 1995. J.M. Napier. Interview with Mr. John M. Napier 1977 and 1983 Mercury Task Force, on January 23 1995. (ChemRisk Repository No. 3270).

NCRP 1985. National Council on Radiation Protection and Measurements. *Radiological Assessment: Predicting the Transport, Bioaccumulation, and Uptake by Man of Radionuclides Released to the Environment*. NCRP Report No. 76. Washington, D.C.

Neathery et al. 1974. M.W. Neathery, W.J. Miller, R.P. Gentry, P.E. Stake, and D.M. Blackmon. Cadmium-109 and methylmercury-203 metabolism, tissue distribution, and secretion into milk of cows. *J. Dairy Sci*. 57(1): 1177-1183.

Ng et al. 1978. Y.C Ng, W.A. Phillips, Y.E. Ricker, R.K. Tandy, and S.E. Thompson. *Methodology for Assessing Dose Commitment to Individuals and to the Population from Ingestion of Terrestrial Foods Contaminated by Emissions from a Nuclear Fuel Reprocessing Plant at the Savannah River Plant*. Lawrence Livermore Laboratory. UCID-17743.

Ng et al. 1979. Y.C. Ng, C.S. Colsher, and S.E. Thompson. Transfer factors for assessing the dose from radionuclides in agricultural products. In: *Proceedings of an International Symposium on Biological Implications of Radionuclides Released from Nuclear Industries, Vienna, 26 April - 3 March 1979.* Vol. II, pp. 295-318. IAEA-SM-237/54 [STI/PUB-522 (V.2)]. International Atomic Energy Agency, Vienna.

Ngim et al. 1992. C.H. Ngim, S.C. Foo, K.W. Boey, and J. Jeyaratnam. Chronic neurobehavioral effects of elemental mercury in dentists. *Br. J. Ind. Med*. 49: 782-790.

Niimi and Kissoon 1994. A.J. Niimi and G.P. Kissoon. Evaluation of the critical body burden concept based on inorganic and organic mercury toxicity to rainbow trout. *Arch. Environ. Contam. Toxicol*. 26: 169-178.

NTP 1993. National Toxicology Program. *NTP Technical Report on the Toxicology and Carcinogenesis Studies of Mercuric Chloride in F344 Rats and B6C3F$_1$ Mice*. U.S. Department of Health and Human Services, Research Triangle Park, NC. NTP TR 408. February 1993.

Oak Ridger 1992. *Y-12 building study a poser- Alpha 4 clean-up challenges DOE.* April 15, 1992. (ChemRisk Repository No. 3265).

Olsen et al. 1990. C.R. Olsen, I.L. Larsen, P.D. Lowry, C.R. Moriones, C.J. Ford, K.C. Dearstone, R.R. Turner, and B.L. Kimmel. *Transport and Accumulation of Cesium-137 and Mercury in the Clinch River and Watts Bar Reservoir System.* ORNL/ER-7. March 1990. (ChemRisk Repository No. 611).

Olsen and Cutshall 1985. C.R. Olsen and N.H. Cutshall. *Contaminant Levels in the Soils and Sediments near the Proposed Blair Road Bridge Construction Site.* Environmental Sciences Division. October 28, 1985.

ORGDP 1981. Oak Ridge Gaseous Diffision Plant (author unknown). *Clinch River and Poplar Creek Bottom Sediments Data Routine Sampling Program 1975-1981.* K-25 Environmental Management Document Center Report No. 00492. (ChemRisk Repository No. 643).

Patterson et al. 1957. G.R. Patterson, C.M. West and J.D. McLendon. *The Y-12 Health Physics Program.* November 1, 1957. Y-1186/del rev. (ChemRisk Repository No. 3261).

Paustenbach et al. 1997. D.J. Paustenbach, G.M. Bruce, and P. Chrostowski. Current views on the oral bioavailability of inorganic mercury in soil: Implications for health risk assesssments. *Risk Anal.* 17(5): 533-544.

Perry and Napier 1957. J.E. Perry and J.M. Napier. *Control of Mercury Vapor in Colex Operations*. November 14, 1957. Y-1185/del rev. (ChemRisk Repository No. 3261).

Piikivi 1989. L. Piikivi. Cardiovascular reflexes and low long-term exposure to mercury vapor. *Int. Arch. Occup. Environ. Health*. 61: 391-395.

Piikivi and Hanninen 1989. L. Piikivi and H. Hanninen. Subjective symptoms and psychological performance of chlorine-alkali workers. *Scand. J. Work Environ. Health*. 15: 69-74.

Piikivi and Ruokonen 1989. L. Piikivi and A. Ruokonen. Renal function and long-term low mercury vapor exposure. *Arch. Environ. Health.* 44(3): 146-149.

Piikivi and Tolonen 1989. L. Piikivi and U. Tolonen. EEG findings in chlor-alkali workers subjected to low long term exposure to mercury vapor. *Br. J. Ind. Med.* 46: 370-375.

Porcella 1994. D.B. Porcella. Mercury in the environment: Geochemistry. Ch. I.1 In: *Mercury Pollution, Integration and Synthesis*. C. J. Watras and J.W. Huckabee, eds. Lewis Publishers, Ann Arbor.

Postma 1971. Letter from H. Postma, Thermonuclear Division UCCND, to J.M. Case, Y-12 Superintendent, regarding mercury vapor in Building 9201-2. January 12, 1971. Y/HG-0183. (ChemRisk Repository No. 3262).

Potter et al. 1972. G.D. Potter, D.R. McIntyre, and G.M. Vattuone. Metabolism of Hg-203 administered as $HgCl_2$ in the dairy cow and calf. *Health Phys*. 22: 13-16.

Prestbo 1996. Letter from Dr. Eric M. Prestbo, Atmospheric Mercury Group Leader, Frontier Geosciences, to Susan Flack of the project team regarding uncertainty associated with use of the portable mercury vapor detector at Y-12. September 5, 1996. (ChemRisk Repository No. 3261).

Reece 1959. J.S. Reece. Letter report to J.S. Reece titled 81-10 Operations on Solvent Contaminated Dirt. May 15, 1959. Y/HG-0499. (ChemRisk Repository No. 3266).

Reece 1974. J.S. Reece. *Preliminary aquatic survey of East Fork Poplar Creek and Bear Creek 1974*. United States Atomic Energy Commission, Oak Ridge Operations. September 20, 1974.

Revis et al. 1989. N.W. Revis, T.R. Osborne, G. Holdsworth, and C. Hadden. Distribution of mercury species in soil from a mercury-contaminated site. *Water, Air, and Soil Pollution*. 45: 105-113.

Richmond and Auerbach 1983. C.R. Richmond and S.I. Auerbach. "Testimony of a Joint Hearing of the Subcommittee on Energy Research and Production and the Subcommittee on Investigations and Oversight of the U.S. Office of Science and Technology Committee on the Impact of Mercury Releases at the Oak Ridge Complex: Summary of Actions and Activities Related to Mercury Releases in the Oak Ridge Area from DOE/UCC-ND Operated Facilities." Included in UCCND (1983b). (ChemRisk Repository No. 448).

Roels et al. 1987. H. Roels, S. Abdeladim, E. Ceulemans, R. Lauwerys. Relationships between the concentrations of mercury in air and in blood and urine in workers exposed to mercury vapour. *Ann. Occup. Hyg*. 31: 135-145.

Rosenman et al. 1986. K.D. Rosenman, J.A. Valciukas, L. Glickman, B.R. Meyers, and A. Cinotti. Sensitive indicators of inorganic mercury toxicity. *Arch. Environ. Health.* 41(4): 208-215.

Ruby et al. 1993. M.V. Ruby, A. Davis, T.E. Link, R. Schoof, R.L. Chaney, G.B. Freeman, P. Bergstrom. Development of an in-vitro screening test to evaluate the in vivo bioaccessibility of ingested mine-waste lead. *Environ. Sci. Technol.* 27(13), 2870-2877.

Ruby et al. 1996. M.V. Ruby, A. Davis, R. Schoof, S. Eberle, and C. Sellstone. "Estimation of lead and arsenic bioavailability using a physiologically based extraction test." *Environ. Sci. Tech.* 39(2), 422-430.

Sager et al. 1984. P.R. Sager, M. Aschner, and P.M. Rodier. Persistent, differential alterations in developing cerebellar cortex of male and female mice after methylmercury exposure. *Dev. Brain Res.* 2: 1-11.

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).

SAIC 1994a. Science Applications International Corporation. *Addendum to the East Fork Poplar Creek – Sewer Line Beltway Remedial Investigation Report*. DOE/OR/2-1119&D2/A1. June 1994. (ChemRisk Repository No. 1826).

SAIC 1994b. Science Applications International Corporation. *Feasibility Study for the East Fork Poplar Creek Sewer Line Beltway*. Volumes 1 and 2. DOE/OR/02-1185&D2&V1 and DOE/OR/02-1185&02&V2. August 1994.

Sanders 1970. *Mercury Analysis of Fish, Water, and Mud Samples Collected in the Oak Ridge Area*. Y/HG-0091/1. August 6, 1970. (ChemRisk Repository No. 1394).

Sapirie 1956. Letter from S.R. Sapirie, Manager, USDOE Oak Ridge Operations, to C.E. Center, Vice-President, Union Carbide, regarding shutdown of Beta-4 plant. March 21, 1956. Y/HG-0535. (ChemRisk Repository No. 3265).

Sapirie 1962. Letter from S.R. Sapirie, Manager, Oak Ridge Operations, to C.E. Arson, Vice-President, Union Carbide, regarding shutdown of Building 9201-4. October 3, 1962. Y/HG-0276. (ChemRisk Repository No. 3265).

Saouter et al. 1995. E. Saouter, M. Gillman, R. Turner, and T. Barkay. Development and field validation of a microcosm to simulate the mercury cycle in a contaminated pond. *Environ. Toxicol. Chem.* 14(1): 69-77.

Scott and Armstrong 1974. D.P. Scott and F.A. Armstrong. Mercury concentration in relation to size in several species of freshwater fishes from Manitoba and northwestern Ontario. *J. Fish. Res. Board Can.* 39: 1685-1690.

Scott 1974. D.P. Scott. Mercury concentrations of white muscle in relation to age, growth, and condition in four species of fishes from Clay Lake, Ontario. *J. Fish. Res. Board. Can.* 31: 1723-1729.

Seafood Safety 1991. *Committee on Evaluation of the Safety of Fishery Products,* Chapter on Methylmercury: FDA Risk Assessment and Current Regulations. National Academy Press, Washington, D.C. p. 196-221.

Shor and Fields 1980. R.W. Shor and D.E. Fields. Agricultural factors affecting the radionuclide foodchain pathway: Green forage consumption by dairy cows. *Health Phys*. 39: 325-332.

Siegel et al. 1974. S.M . Siegel, N.J. Puerner, and T.W. Speitel. Release of volatile mercury from vascular plants. *Physiologia Pl.* 32: 174-176.

Skerfving 1988. S. Skerfving. Mercury in women exposed to methylmercury through fish consumption, and in their newborn babies and breast milk. *Bull. Environ. Contam. Toxicol.* 41: 475-482.

Skidmore Owings & Merrill 1948. Report to the Atomic Energy Commission on the Master Plan, Oak Ridge Tennessee. Prepared by Skidmore Owings & Merrill. December.

Smith 1944. Memorandum from S.B. Smith, Industrial Hygiene, regarding an investigation of mercury use in Y-12 Building 9202. July 20, 1944. (ChemRisk Repository No. 3265).

Smith 1966. Memorandum from D.W. Smith, Union Carbide, to R.D. Williams, Union Carbide, regarding mercury in the Building 9201-5 fan room. April 12, 1966. Y/HG-0070. (ChemRisk Repository No. 3265).

Smith et al. 1970. R.G. Smith, A.J. Vorwald, L.S. Patil, and T.F. Mooney. Effects of exposure to mercury in the manufacture of chlorine. *Am. Ind. Hyg. Assoc. J.* Nov-Dec. 687-700.

Steffek et al. 1987. A.J. Steffek, R. Clayton, C. Siew, and A.C. Verrusio. Effects of elemental mercury vapor exposure on pregnant Spraque-Dawley rats. *Teratology* 35:59A.

Stein et al. 1996. E.D. Stein, Y. Cohen, and A.M. Winer. Environmental distribution and transformation of mercury compounds. In: *Critical Reviews in Environmental Science and Technology*. T.R. Logan, ed. 26(1):1-43.

Stern 1993. A.H. Stern. Re-evaluation of the reference dose for methylmercury and assessment of current exposure levels. *Risk Analysis.* 13(3): 355-364.

Stevens 1991. J.B. Stevens. Disposition of toxic metals in agricultural food chain. 1. Steady-state bovine milk biotransfer factors. *Environ. Sci. Technol*. 25(7): 1289-1294.

Stiff 1982. A.C. Stiff. Poplar Creek Fish Sampling Data: Special Sampling Program 1982 Only. *K-25 Environmental Management Document Center Report No. 00491*. October 8, 1992. (ChemRisk Repository No. 635).

Stonard et al. 1983. M.D. Stonard, B.V. Chater, D.P. Duffield, A.L. Nevitt, J.J. O'Sullivan, and G.T. Steel. An evaluation of renal function in workers occupationally exposed to mercury vapour. *Int. Arch. Occup. Environ. Health.* 52: 177-189.

Sykes 1981. J.B. Sykes. Letter from J.B. Sykes to J.D. McLendon. *Radiation Safety Records on Storage in the Y-12 Records Center*. October 6, 1981. (ChemRisk Repository No. Task 5 Notebook).

Taylor 1989. F.G. Taylor, Jr. *Mercury Assessment for Water and Sediment in Oak Ridge National Laboratory Streams*. March 1989. ORNL/M-713. (ChemRisk Repository No. 3269 or 163).

TDHE 1983. Tennessee Department of Health and Environment. *Miscellaneous Correspondence to Oak Ridge Community Concerning DOE Mercury Analyses in the Oak Ridge Vicinity*. January–November 1983.

Thomason and Associates 1996. *Historic Building Assessment of the Oak Ridge Y-12 Plant- DRAFT*. May 1996. Y/TS-1525. (ChemRisk Repository No. 3267).

Todd 1990. R.M. Todd. *Commercial Fishing Survey 1989.* Tennessee Wildlife Resources Agency, Nashville, TN. November.

Travis et al. 1989. C.C. Travis, B.G. Blaylock, K.L. Daniels, C.S. Gist, F.O. Hoffman, R.J. McElhaney, and C.W. Weber. *Final Report of the Oak Ridge Task Force Concerning Public Health Impact of the Off-Site Contamination in East Fork Poplar Creek and Other Area Streams.* ORNL/TM-11252. August 1989. (ChemRisk Repository No. 168).

Turner and Bloom 1995. Reconstruction of historical atmospheric Hg releases using analysis of tree rings in red cedar (Juniperus virginia). Poster presentation, Oak Ridge National Laboratory.

Turner and Bogle 1992. R.R. Turner and M.A. Bogle. *Mercury in Ambient Air over Floodplain of East Fork Poplar Creek, Oak Ridge, Tennessee*. Oak Ridge National Laboratory. July.

Turner et al. 1984. R.R. Turner, C.R. Olsen, and W.J. Wilcox, Jr. Environmental fate of Hg and 137Cs discharged from Oak Ridge facilities. *Trace Subst. Environ. Health*. 18: 329-338.

Turner 1990. R.R. Turner. *Review of Available Information for Mercury at Building 9201-2*. February 1990. Y/TS-626. (ChemRisk Repository No. 3277).

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. 1993. R.R. Turner, T. Barkay, and E. Saouter. Bioreduction of mercury in contaminated effluents. Preprint extended abstract, presented at the I&EC Special Symposium, American Chemical Society, Atlanta, GA. September 27-29, 1993.

TVA 1959. Tennessee Valley Authority. *Floods on Clinch River & East Fork Poplar Creek in Vicinity of Oak Ridge, Tennessee*. Division of Water Control Planning, Knoxville, Tennessee. Report No. 0-5922. September 1959.

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: Water Sampling and Analysis*. Office of Natural Resources and Economic Development, Knoxville, Tennessee. January 1985. (ChemRisk Repository No. 842).

TVA 1985d. Tennessee Valley Authority. *Instream Contaminant Study, Task 3: Sediment Transport*. Office of Natural Resources and Economic Development, Knoxville, Tennessee. August 1985. (ChemRisk Repository No. 843).

TVA 1985e. 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 1987. Tennessee Valley Authority. *Estimation of the Bioaccumulation of Mercury by Bluegill Sunfish in East Fork Poplar Creek: Final Report for the Period 1 October 1985 – 30 April 1987*. Office of Natural Resources and Economic Development, Knoxville, Tennessee. April 1987. (ChemRisk Repository No. 853).

TVA 1988. Tennessee Valley Authority. *Surface Water Monitoring Strategy– Ambient Monitoring – Results from Analysis of Fish Tissue Collected in 1986.* Office of Natural Resources and Economic Development, Knoxville, Tennessee.

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. (ChemRisk Repository No. 845).

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).

TVA 1991a. Tennessee Valley Authority. *Reservoir Monitoring– 1990. Fish Tissue Studies in the Tennessee Valley in 1989.* Office of Water Resources, Chattanooga, Tennessee. TVA/WR/AB-91/12. October 1991. (ChemRisk Repository No. 847).

TVA 1991b. Tennessee Valley Authority. *Results of Sediment and Water Sampling for Inorganic, Organic, and Radionuclide Analysis at Recreation Areas and Water Intakes– Norris, Melton Hill, and Watts Bar Lakes: Data Report*. Water Quality Department. (ChemRisk Repository No. 852).

TWRA 1993. *Reports on Recreational Fishing at Watts Bar and Melton Hill Reservoirs.* Tennessee Wildlife Resources Agency, Nashville, TN. Received February, 1993.

UCC 1972. Union Carbide Corporation. *Environmental Monitoring Report, United States Atomic Energy Commission, Oak Ridge Facilities. Calendar Year 1971.* UCC-ND-221. (ChemRisk Repository No. 942).

UCC 1973. Union Carbide Corporation. *Environmental Monitoring Report, United States Atomic Energy Commission, Oak Ridge Facilities. Calendar Year 1972.* 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.* 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.* 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*. Y/UB-4. (ChemRisk Repository No. 946).

UCC 1977. Union Carbide Corporation. *Environmental Monitoring Report, United States Energy Research and Development Administration, Oak Ridge Facilities. Calendar Year 1976*. 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*. 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*. 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.* 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*. 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.* Y/UB-16. (ChemRisk Repository No. 952).

UCCND 1983a. 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– 1950-1983*. Y/EX-21/del rev. August 18, 1983. (ChemRisk Repository No. 449).

UCCND 1983b. Union Carbide Company Nuclear Division. The 1983 Mercury Task Force. *Mercury at the Y-12 Plant: A Summary of the 1983 UCCND Task Force Study*. Y/EX-23. November 1983. (ChemRisk Repository No. 448).

UCCND 1983c. Classified version of UCCND 1983a. Y/EX-21. August 18, 1983.

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).

USAEC 1966. United States Atomic Energy Commission. *USAEC Report of Investigating Committee Loss of Mercury at the Y-12 Plant*. May 13, 1966. Part 1-Y/HG-0072/del rev. (ChemRisk Repository No. 3259).

USDA 1955a. U.S. Department of Agriculture. *Dietary Levels of Households in the South, Household Food Consumption Survey 1955, Report No. 9*. Washington, D.C.

USDA 1955b. U.S. Department of Agriculture. *Dietary Levels of Households in the United States, Household Food Consumption Survey 1955, Report No. 6*. Washington, D.C.

USDA 1966. U.S. Department of Agriculture. *Food Consumption of Households in the United States, Seasons and Year 1965-1966, Report No. 12*. Washington, D.C.

USDOE 1993. United States Department of Energy. *Openness Press Conference Fact Sheets*. December 7, 1993. (ChemRisk Repository No. 3265).

USDOI. 1993. U.S. Department of the Interior. *1991 National Survey of Fishing, Hunting, and Wildlife-Associated Recreation - Tennessee*. U.S. Department of the Interior, Fish and Wildlife Service, and U.S. Department of Commerce, Bureau of the Census. July.

USEPA 1985. United States Environmental Protection Agency. *Development of Statistical Distributions or Ranges of Standard Factors used in Exposure Assessments*. USEPA/600-8-85-010. Prepared for USEPA, Washington, D.C., by GCA Corporation, Chapel Hill, NC.

USEPA 1988. United States Environmental Protection Agency. *Drinking Water Criteria Document for Inorganic Mercury (Final)*. Cincinnati, OH. PB89-192207. July, 1988.

USEPA 1989a. United States 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. USEPA540/1-89/002. December 1989.

USEPA 1992. United States Environmental Protection Agency. *Dermal Exposure Assessment: Principles and Applications*. Office of Research and Development. USEPA/600/8-91/011B. January 1992.

USEPA 1995a. United States Environmental Protection Agency. *Exposure Factors Handbook*. Exposure Assessment Group, Office of Health and Environmental Assessment. Washington, D.C. USEPA/600/P-95/002A. June 1995.

USEPA 1995c. United Stated Environmental Protection Agency. *User's Guide for the Industrial Source Complex (ISC) Dispersion Models.* Research Triangle Park, North Carolina, March. ISCST3 version 96113. USEPA-454/B-95-003.

USEPA 1997. United States Environmental Protection Agency. Mercury Study Report to Congress Volumes I through 5. Office of Air Quality Planning and Standards and Office of Research and Development. December.

USGS 1952. United States Geological Survey. Clinton Quadrangle, Clinton Tennessee. 7.5 Minute Series (Topographic) Map for 1952.

USGS 1953a. United States Geological Survey. Bethel Valley Quadrangle, Bethel Valley, Tennessee. 7.5 Minute Series (Topographic) Map for 1953a.

USGS 1953b. United States Geological Survey. Windrock Quadrangle, Windrock, Tennessee. 7.5 Minute Series (Topographic) Map for 1953a.

USGS 1967. United States Geological Survey. *Hydrologic Data for the Oak Ridge Area Tennessee.* Geological Survey Water-Supply Paper 1839-N. United States Government Printing Office, Washington, DC.

USGS 1968a. United States Geological Survey. Bethel Valley Quadrangle, Bethel Valley, Tennessee. 7.5 Minute Series (Topographic) Map for 1968.

USGS 1968b. United States Geological Survey. Clinton Quadrangle, Clinton Tennessee. 7.5 Minute Series (Topographic) Map for 1968.

USGS 1975. United States Geological Survey. Clinton Quadrangle, Clinton Tennessee. 7.5 Minute Series (Topographic) Map for 1968, photorevised in 1975.

USGS 1989. United States Geological Survey. Bethel Valley Quadrangle, Bethel Valley, Tennessee. 7.5 Minute Series (Topographic) Map for 1989.

USGS 1990. United States Geological Survey. Clinton Quadrangle, Clinton Tennessee. 7.5 Minute Series (Topographic) Map for 1968, photorevised in 1990.

U.S. Weather Bureau 1953. *A Meteorological Survey of the Oak Ridge Area: Final Report Covering the Period 1948 - 1952. U.S. Atomic Energy Commission, Oak Ridge Operations*. Oak Ridge, Tennessee. Report ORO-99. November 1953. (ChemRisk Repository No. 1211).

Van Winkle et al. 1984. W. Van Winkle, R.W. Counts, J.G. Dorsey, J.W. Elwood, V.W. Lowe, Jr., R. McElhaney, S.D. Schlotzhauer, F.G. Taylor, Jr., and R.R. Turner. *Mercury Contamination in East Fork Poplar Creek and Bear Creek*. ORNL/TM-8894. February 1984. (ChemRisk Repository No. 171).

Verberk et al. 1986. M.M. Verberk, H.J.A. Salle, and C.H. Kemper. Tremor in workers with low exposure to metallic mercury. *Am. Ind. Hyg. Assoc. J.* 47(8): 559-562.

Vreman et al. 1986. K. Vreman, N.G. van der Veen, E.J. van der Molen, and W.G. de Ruig. Transfer of cadmium, lead, mercury, and arsenic from feed into milk and various tissues of dairy cows: Chemical and pathological data. *Neth. J. of Ag. Sci*. 34: 129-144.

Westoo 1973. G. Westoo. Methylmercury as a percentage of total mercury in flesh and viscera of salmon and sea trout of various ages. *Science.* 181: 567-568.

Whicker and Kirchner 1987. F.W. Whicker and T.B. Kirchner. Pathway: A dynamic food-chain model to predict radionuclide after fallout deposition. *Health Phys*. 52(6): 717-737.

WHO 1976. World Health Organization. Environmental Health Criteria: Mercury. Geneva, Switzerland.

Wilcox 1995. W.J. Wilcox, Jr. Interview with Mr. William J. Wilcox, Jr. 1983 Mercury Task Force Chairman, on January 23, 1995. (ChemRisk Repository No. 3270).

Wren 1996. C. Wren. *Review of Mercury Levels in Fish, Water, and Sediments. Progress Report #2 for Oak Ridge Dose Reconstruction Project*. Ecological Services for Planning, Ltd. Guelph, Ontario, Canada. August.

Xiao et al. 1991. Z.F. Xiao, J. Munthe, W.H. Schroeder, and O. Lindqvist. Vertical fluxes of volatile mercury over forest soil and lake surfaces in Sweden. *Tellus*, 43B. 267-279.

Young 1991. R.A. Young. *Toxicity Summary for Mercury*. Health and Safety Research Division, Oak Ridge National Laboratory. November.

Zitko et al. 1971. V. Zitko, B.J. Finlayson, D.J. Wildish, J.M. Anderson, and A.C. Kohler. Methylmercury in freshwater and marine fishes in New Brunswick, in the Bay of Fundy, and on the Nova Scotia Banks. *J. Fish. Res. Board. Can.* 28: 1285-1291.

This page intentionally left blank.

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