REPORTS OF THE OAK RIDGE DOSE RECONSTRUCTION, Vol. 4
The Report of Project Task 4 • July 1999

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

Submitted to the Tennessee Department of Health by

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OAK RIDGE HEALTH STUDIES
OAK RIDGE DOSE RECONSTRUCTION

– TASK 4 REPORT –

RADIONUCLIDES RELEASED 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

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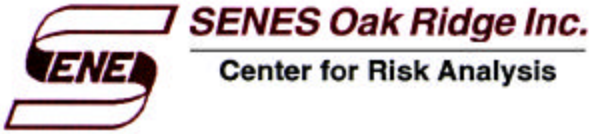

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TABLE OF CONTENTS

APPENDICES (Volume 4A)

The numerals that occur before the letters in the appendix titles indicate the section of the main report text to which they are related. Not all sections have appendices.

LIST OF FIGURES

FIGURES IN APPENDICES (Volume 4A)

The numerals that occur before the letters in the appendix titles indicate the section of the main report text to which they are related. Not all sections have appendices.

LIST OF TABLES

TABLES IN APPENDICES (Volume 4A)

The numerals that occur before the letters in the appendix titles indicate the section of the main report text to which they are related. Not all sections have appendices.

EXECUTIVE SUMMARY

Purpose

The purposes of Task 4 of the Oak Ridge Dose Reconstruction were (1) to estimate the historical releases from the X-10 facility to the Clinch River, (2) to evaluate the potential pathways by which members of the public could have been exposed to radioactive effluents in the Clinch River between 1944 and 1991, and (3) to calculate radiation doses and risks to reference individuals who were potentially exposed to radioactivity released to the Clinch River from the X-10 facility. Direct measurement of the amounts of radionuclides taken up by the organs of specific individuals since 1944 is no longer feasible because most of these radionuclides have short residence times in the human body. Therefore, a dose reconstruction has been necessary to determine the magnitude and extent of past exposure and to interpret the health consequences of these exposures. This dose reconstruction relies upon independent evaluation of the amounts of radionuclides released, reported environmental measurements, and mathematical models to estimate the magnitude and extent of past exposures, doses, and health risks.

Background

In the early days of the Manhattan Project, the Clinton Laboratory, later referred to as the X-10 facility and now called the Oak Ridge National Laboratory, was designed to operate for one year as a pilot plant for the Hanford, Washington, operations. All radioactive wastes generated from this facility were to be stored in large underground "gunite" tanks. The original plans changed, and in 1944 the first radioactive effluents from the X-10 site entered White Oak Creek and flowed into White Oak Lake. White Oak Lake served as the final settling basin for contaminants released to White Oak Creek. Radionuclides remaining in the water column were released from the X-10 site with the flow of water from White Oak Dam, which is located 1 km (0.6 mile) upstream from the Clinch River (see map, p. 7-2).

Sources of Radioactive Waste

During the early years of X-10 operations, the graphite reactor and the "hot pilot plant" (a chemical separation plant) were the major sources of radioactive wastes. Wastes from the "hot pilot plant" were placed in open waste pits; high levels of ^{106}Ru began seeping from the pits into White Oak Lake in 1959. Strontium-90 and ^{137}Cs had also been placed in the pits, but these isotopes were retained by nearby soils; however, amounts of ^{106}Ru as high as 7.4×10^{13} Bq (2000 curies) per year were released from White Oak Dam from 1959 to 1963. From 1944 to 1991, approximately 5.9×10^{15} Bq (160,000 curies) of radioactivity were released over White Oak Dam to the Clinch River; of this amount, 91% was tritium, and the rest was mixed fission and activation products.

Evidence suggests that a secondary source of radionuclides released to the Clinch River was the scouring of contaminated sediment from White Oak Creek Embayment. After White Oak Lake was drained in 1955, heavy rainfall scoured the bottom sediment of White Oak Lake, resulting in the deposition of particle reactive radionuclides (primarily ^{137}Cs) in White Oak Creek Embayment. The peaking discharges from Melton Hill Dam, which was completed in 1963, resulted in the backflow of

water up White Oak Creek Embayment and the scouring of radionuclide-containing sediments into the Clinch River. A coffer cell dam was constructed at the mouth of White Oak Creek in 1990 to prevent the backflow of water up White Oak Creek Embayment, and scouring of embayment sediment ceased at that time.

Screening Analysis

To focus time and resources on the radionuclides that were most likely to have been important in terms of dose or risk to off-site individuals, a conservative screening evaluation was conducted. Twenty-four radionuclides released into the Clinch River from the X-10 site from 1944 to 1991 were considered as potential contaminants of concern. The conservative screening analysis identified those radionuclides and pathways for which the human health risk was clearly below a minimum level of concern. Nine exposure pathways and sixteen radionuclides, including tritium, were identified as low priority for further consideration because conservative screening estimates were at least a factor of ten below the screening guide value of one chance in ten thousand (1×10^{-4}) of excess incidence of disease, as established by the Oak Ridge Health Agreement Steering Panel (ORHASP). Of the eight remaining radionuclides, ^{137}Cs, ^{60}Co, ^{106}Ru, and ^{90}Sr were expected to be the most important contributors to radiological dose and subsequent excess health risk.

Radionuclides Released from White Oak Dam

The dose reconstruction relies on estimates and reported measurements of radionuclides released from White Oak Dam from 1944-1991. A detailed investigation was performed for (1) the methods used for measurements of radioactive releases from White Oak Dam, (2) the methods used for estimation of flow rates at White Oak Dam, and (3) the uncertainties associated with these measurements. Estimates of the quantities of radionuclides historically released from White Oak Dam were based on laboratory documents, available log books, and interviews with personnel who were responsible for, or involved in, the collection of samples and monitoring of radioactive releases at White Oak Dam. Direct measurements of the radionuclides released from White Oak Dam were available, except for the years 1944 to 1949. For these years, estimates were based on the fraction that each radionuclide contributed to a measurement or estimate of gross beta activity. Detailed source terms (annual release amounts) were developed for the following radionuclides: ^{60}Co, ^{90}Sr, ^{95}Nb, ^{95}Zr, ^{106}Ru, ^{131}I, ^{137}Cs and ^{144}Ce. The uncertainty of the source terms varied over time because of various changes in sampling and analytical methods and waste disposal or treatment events.

Estimated Radionuclide Concentrations in Water and Sediments

Measured concentrations of radionuclides in water are available for many years for several locations downstream from the confluence of White Oak Creek and the Clinch River (CRM 20.8). These measurements were not entirely consistent as to location or method of measurement and did not include all the radionuclides of concern. Therefore, a modeling effort was conducted to estimate the annual average concentrations of radionuclides in water at specific locations downstream of White Oak Creek. A modified version of the HEC-6 aquatic transport model (HEC-6-R) was used to estimate historical water concentrations. The annual average releases of specific radionuclides from White Oak Dam were used in the modeling analysis. The uncertainty of the modeled water concentrations was much higher

than the uncertainty about water concentrations obtained from measurements; therefore, measurements for specific locations and time periods were used in preference to model predictions when the data were sufficient to estimate an annual average concentration in water. In particular, the model did not always account sufficiently for localized scouring of sediment after Melton Hill Dam began operation in 1963.

Estimated shoreline concentrations of radionuclides in sediment were obtained by using the HEC-6-R model to track the sediment inventory in various reaches of the Clinch River. Monitoring data collected in the 1990s were used to calibrate the shoreline sediment estimates. Because of the limited data, all shoreline sediment concentrations used in the risk assessment were based on model estimates.

Estimation of Exposure to Reference Individuals

For all locations, the exposure pathways of interest include fish ingestion and ingestion of milk and meat; other exposure pathways of interest varied with location. For the Jones Island area (CRM 21.0 to 17.0), the exposure pathways of interest were fish ingestion, external exposure from shoreline sediment, and ingestion of meat and milk. The exposure pathways for the K-25/Grassy Creek area (CRM 17.0 to CRM 5.0) included fish ingestion, drinking water, external exposure to shoreline sediment, and ingestion of milk and meat. For the Kingston Steam Plant area (CRM 5.0 to CRM 2.0), the important pathways were fish ingestion, external exposure to shoreline sediment, drinking water, and ingestion of milk and meat. Exposure pathways for residents of the City of Kingston (CRM 2.0 to CRM 0.0) included fish ingestion, external exposure to shoreline sediment, drinking water, and ingestion of milk and meat from livestock having direct access to the river as a source of drinking water.

Reference individuals in this study were identified with respect to the pathways involved, the specific characteristics of the individual pathways, and the size and type of the population affected. For the fish pathway, reference individuals were defined in terms of fish consumption rates as Category I (people who consumed fish on a regular basis, i.e., 1 to 2.5 meals per week or 7.1-33 kg y^{-1} for males and 5.7-27 kg y^{-1} for females), Category II (0.25-1.3 meals per week or 2.2-16 kg y^{-1} for males and 1.8-14 kg y^{-1} for females), or Category III (0.04-0.33 meals per week or 0.39-4.3 kg y^{-1} for males and 0.32-3.6 kg y^{-1} for females).[1] (Meal size was defined as 0.10-0.30 kg per meal for males and 0.08-0.25 kg per meal for females. The ranges shown are the 95% subjective confidence intervals and do not include the extreme values on either end.) For all categories, it was assumed that 20-100% of the fish was contaminated and that 80-90% of the radioactivity in the fish was retained after processing.

Two reference individuals, an adult and a child, were used for the water ingestion pathway. Children were not considered for the K-25/Grassy Creek area or the Kingston Steam Plant area, because these are industrial facilities and it is not likely that children would have obtained drinking water from these locations. However, both children and adults were exposed via the City of Kingston water supply. Multiple reference individuals were considered for the milk ingestion pathway, including children who could have consumed different amounts of home-produced milk depending on whether they were at home or in school. Adults were considered as the reference individuals for both the meat ingestion pathway and the external exposure pathway External exposure calculations were based on shoreline

[1] These categories correspond to the following ranges (95% subjective confidence intervals) in pounds: Category I, 16-72 lb y^{-1} for males and 13-60 lb y^{-1} for females; Category II, 4.8-36 lb y^{-1} for males and 3.9-30 lb y^{-1} for females; Category III, 0.85-9.4 lb y^{-1} for males and 0.70-7.9 lb y^{-1} for females.

usage of 75-430 h y^{-1} at CRM 20.5, 85-440 h y^{-1} at CRM 14, or 130-490 h y^{-1} at CRM 3.5 or CRM 0 (95% subjective confidence intervals).

Estimation of Organ-Specific Radiation Doses

The International Commission on Radiological Protection (ICRP) has developed a methodology to calculate internal radiation doses to people ingesting contaminated food or drinking contaminated water. To account for the uncertainties introduced by variability among individuals, a range of values was developed for the factors that specify the dose per unit intake for a given radionuclide. To obtain the ranges of possible values for ^{137}Cs, ^{60}Co, and ^{106}Ru, the published ICRP ingestion dose factors were modified by application of multiplicative uncertainty factors, the values of which were dependent on the radionuclide and organ. In addition, new dose conversion factors and associated uncertainties were calculated for ^{90}Sr and ^{131}I, based on the ICRP methodology. Dose conversion factors were derived for all internal organs of importance. A sensitivity analysis was performed to determine which of the biokinetic parameters contribute the most to the uncertainty in the dose conversion factors. Each dose conversion factor was specified as a range of values rather than a point estimate.

Fish Ingestion

The estimated organ doses to individuals consuming fish exceeded the dose estimates for all other pathways. The highest doses were for Category I consumers of fish (1-2.5 meals per week) at CRM 20.5, just below the confluence of White Oak Creek and the Clinch River. Central values of the cumulative doses for 1944 to 1991 for specific organs ranged from 0.31 (skin) to 0.81 cSv (bone) for males and from 0.23 (skin) to 0.60 cSv (bone) for females (1 cSv equals 1 rem); the 95% subjective confidence intervals ranged from about 0.02 to 8 cSv. Organ doses were generally lower for females than for males, due to the lower ingestion rate assumed for females. For Category I consumers of fish near the city of Kingston (CRM 0), the organ doses are about a factor of 8-9 lower than those estimated for CRM 20.5. Estimated organ doses for Category II and III consumers of fish are lower than those for Category I in proportion to the lower intake rates assumed for these categories of individuals.

Other Exposure Pathways

Organ-specific doses from external exposure were about a factor of 1.1-3.5 lower than the doses to a Category I consumer of fish at CRM 14, with the largest doses to skin, bone, and thyroid. Adults who spent time along the shoreline but who seldom consumed fish probably received the same or higher organ doses from external exposure as from fish ingestion.

For most organs, doses from drinking water at CRM 14 and CRM 3.5 were lower than the doses from external exposure at the same location. However, for the large intestine, bone, and red bone marrow, the doses from drinking water were higher than those from external exposure or consumption of fish (by Category II or III consumers) due to the presence of ^{90}Sr and ^{106}Ru.

Estimated doses from ingestion of meat and milk were lower than those for ingestion of drinking water by 1 to 3 orders of magnitude. The highest doses were to the large intestine, bone, red bone marrow, and (for the ingestion of milk) the thyroid.

Estimates of Thyroid Dose to a Child from the Drinking Water and Milk Ingestion Pathways

The 95% subjective confidence intervals for the estimated doses to a child 0 to 14 years of age drinking home-produced milk at CRM 14 or CRM 3.5 from 1946-1960 were 0.00058 to 0.054 cSv (0.0062 central value) and 0.00055 to 0.042 cSv (0.0044 central value), respectively. The 95% subjective confidence interval for the estimated drinking water dose for a child living in Kingston (CRM 0.0) was 0.000039 to 0.0021 cSv (0.00031 central value), and for the combined pathways (drinking water and milk), 0.00014 to 0.0047 cSv (0.00091 central value). The exposure period for a child drinking water or water and milk was different from that for drinking milk alone because the Kingston City municipal water supply did not become a potential source of contamination until 1955.

Estimates of Excess Lifetime Risk of Cancer Incidence

The organ-specific dose estimates were used as the basis for organ-specific and total estimates of excess lifetime risk of cancer incidence. The dose-response functions were based on cancer incidence data from the Japanese A-bomb survivors, the background incidence of cancer for East Tennessee, and the use of relative versus absolute risk models to transfer epidemiological findings in the A-bomb survivors to populations exposed to radionuclides released to the Clinch River. The uncertainty due to differences between exposures at high and low dose rates was considered explicitly in the calculation of risk for each organ. Extension of the calculations from dose to risk accounts for differing radiosensitivity among organs and permits identification of the most important target organs. In addition, estimation of the risks facilitates direct interpretation of the exposures in terms of their potential impact on people's health.

Fish Ingestion

For Category I consumers of fish (1-2.5 meals per week) near Jones Island (CRM 20.5), the 95% subjective confidence interval of the total excess lifetime risk of cancer incidence for all radionuclides and organs was 3.6×10^{-5} to 3.5×10^{-3} (central value, 2.8×10^{-3}) for males and 2.9×10^{-5} to 2.8×10^{-3} (central value, 2.3×10^{-4}) for females. The difference in risk between males and females primarily reflects the difference in ingestion rates. For both males and females, the largest contribution to the total risk (about 90%) is from ^{137}Cs.

For any given location, risks of excess lifetime cancer incidence for Category I consumers of fish are greater than those for Category II and III consumers by factors of about 2 and 8, respectively, in proportion to the different intake rates assumed for these reference individuals (Figure ES.1). The upper bound on the total risk from fish consumption for Category I or II consumers (1-2.5 or 0.25-1.3 meals per week, respectively) reaches or exceeds 1×10^{-4} at all locations (Figure ES.2); central values exceed 1×10^{-4} only for Category I and II consumers at CRM 20.5. For Category III consumers (0.04-0.33 meals per week), the upper 95% subjective confidence limit on the total risk estimate is below 1×10^{-4} for all locations except CRM 20.5.

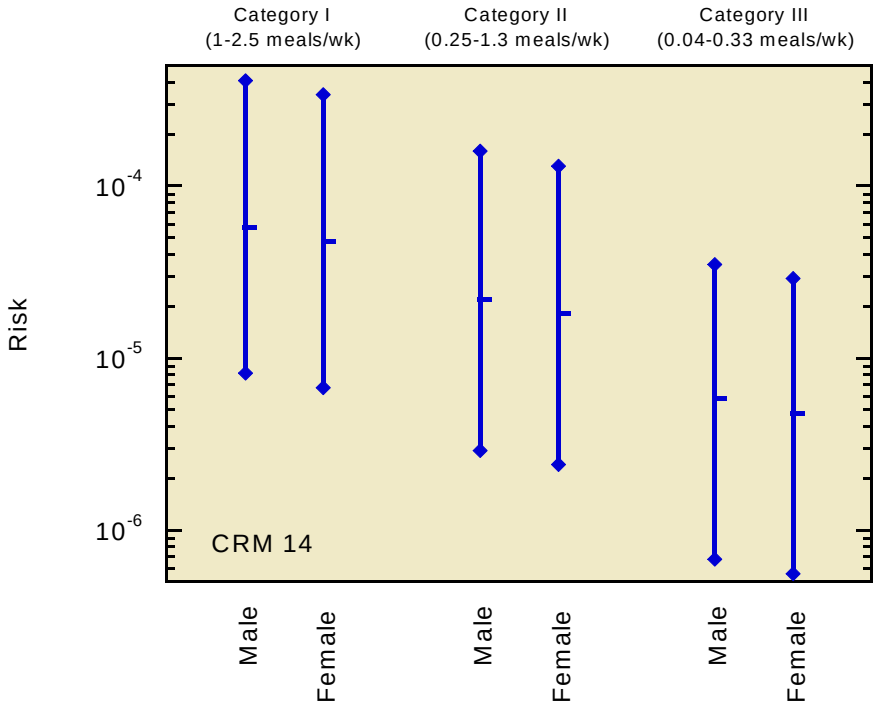

Figure ES.1 Excess lifetime risk of cancer incidence for males and females consuming fish at CRM 14. The vertical lines indicate the 95% subjective confidence intervals of the estimated risks; the central values (50th percentiles) are also indicated.

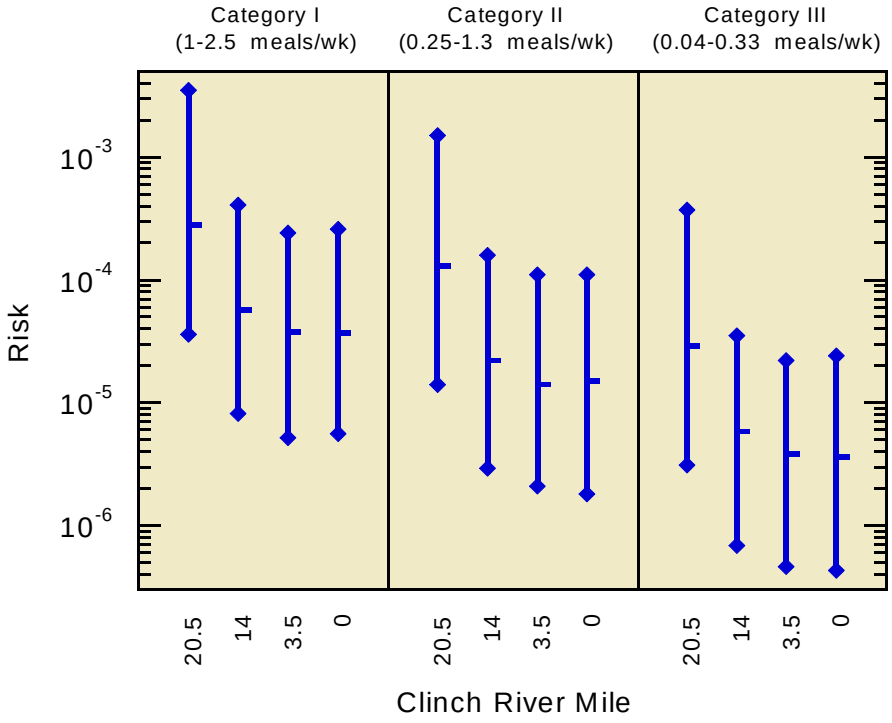

Figure ES.2 Excess lifetime risk of cancer incidence from all exposure pathways for three reference males consuming fish at different rates at each location of interest along the Clinch River. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; central values (50th percentiles) are also indicated. Risks for females are slightly lower than for males (see Fig. ES.1).

For ingestion of fish from the Jones Island area (CRM 20.5), the upper bounds on the risk for both males and females exceeded 1×10^{-4} for bladder, stomach, lower large intestine, lungs, and red bone marrow (leukemia), as did the upper bounds on the risk estimates for breast in females and for liver in males. Although the breast received among the lowest doses of any organ, the breast has the highest risk of all the organs examined (upper bound, 9.3×10^{-4}). The highest risk for males and second highest risk for females is for the red bone marrow (upper bounds of 3.4×10^{-4} and 4.0×10^{-4}, respectively). The difference between the highest and lowest organ-specific risks at any one location is about a factor of 70-80 for females and 40 for males, although the differences in doses were only a factor of 2-4. This situation illustrates the great difference in organ sensitivities to radiation-induced cancer and underlines the importance of calculating risks as well as doses in a dose reconstruction study, because the organ with the highest dose may not be the organ at highest risk.

For individuals using or residing on Watts Bar Reservoir, the exposures, doses, and risks are substantially lower than they are for individuals using any segment of the Clinch River. Our best estimate is that exposures from the past consumption of contaminated fish in Watts Bar Reservoir are 4 to 25 times less than for persons catching fish from the Clinch River near the K-25/Grassy Creek area (CRM 14), assuming similar ingestion rates.

Other Exposure Pathways

Depending on the location, the external dose from shoreline sediments (based on exposure of approximately 100-500 h y^{-1}, depending on location) contributes as much as 90% of the total risk from all pathways for a Category II consumer of fish (0.04-0.33 meals per week); fish ingestion contributes about 10%, and drinking water from 2 to 30% of the total risk of cancer incidence. For Category II consumers of fish (0.25-1.3 meals per week), fish ingestion contributes 30-40% of the total risk, depending on location, and for Category I (1-2.5 meals per week), about 50-60%, except for CRM 20.5, where the external exposure is low and exposure via drinking water did not occur. For the external exposure pathway alone, the upper bounds at all locations except CRM 20.5 barely reach 1×10^{-4} (highest value, 1.2×10^{-4} at CRM 0), indicating a low likelihood that this level was actually exceeded; for drinking water alone, the upper bound at all locations is below 1×10^{-4} (highest value, 4.6×10^{-5} at CRM 14). For the combined pathways at CRM 20.5, the upper bounds on the total excess lifetime risk were 3.6×10^{-3}, 1.7×10^{-3}, and 4.1×10^{-4} for male consumers of fish in Categories I, II, and III, respectively. For the other locations, the highest upper bound values were 5.9×10^{-4}, 3.4×10^{-4}, and 1.9×10^{-4} for male consumers of fish in Categories I, II, and III, respectively, all at CRM 14.

Estimates Of Excess Lifetime Risk of Thyroid Cancer for a Child from the Drinking Water and Milk Ingestion Pathways

The highest excess lifetime risk of thyroid cancer occurred for a female child ingesting milk obtained from an area near CRM 14 between 1946 and 1960 (95% confidence interval, 1.1×10^{-7} to 2.5×10^{-5}; central value, 1.8×10^{-6}). The 95% subjective confidence interval on the risk for a female child exposed via the combined drinking water and milk ingestion pathways (milk from CRM 3.5 and water from CRM 0, between 1955 and 1969) was 2.4×10^{-9} to 1.8×10^{-6} (central value, 2.4×10^{-7}).

Risk Estimates for Shorter Exposure Periods

In most cases, individuals were not exposed to the various pathways over the entire period from 1944 to 1991. In addition, both the operations at the X-10 site and the releases of radionuclides to the Clinch River changed over time. To account for more realistic exposure times, risks were summarized by decade. The first two decades (1944-1953 and 1954-1963) produced the highest risks for each pathway and from all pathways combined (shown for males in Figure ES.3; values for females are slightly lower for the fish ingestion pathway). In the first decade, the ingestion of fish dominated the total risk; however, external exposure to shoreline sediments became increasingly important in later years. Because the ingestion of fish and external exposure to shoreline sediments contribute most of the excess lifetime risk of cancer incidence, ^{137}Cs is the dominant radionuclide in all decades. In addition to risk estimates by decade, estimates of total risk per year at CRM 14 were also made in terms of risk per pound of fish consumed, per hour exposed to shoreline sediment, and per liter of water consumed.

Contribution to Uncertainty in the Risk Estimates

This study explicitly included uncertainty in external dosimetry, internal dosimetry, and dose-response relationships (risk factors), as well as uncertainty in the various parameters affecting the exposure estimates. For all locations and ingestion rates examined, the dominant sources of uncertainty in the risk from fish ingestion are the concentration of ^{137}Cs in fish and the amount of fish consumed. The relative importance of a specific parameter depends on the location of exposure and the ingestion rates; in most cases, the bioconcentration factor is the single most important parameter affecting the overall uncertainty. For external exposure, the most important contributors to uncertainty are the concentrations of ^{137}Cs and ^{60}Co in shoreline sediments, followed by the total exposures and the risk factors. For internal exposure via drinking water, the most important sources of uncertainty are the amount of contaminant consumed, followed by the risk factors and the concentrations of ^{106}Ru and ^{90}Sr in the water. Uncertainty in dosimetry contributes less than 5% (internal) or 10% (external) of the total uncertainty, while the risk factor (except for internal exposure to ^{137}Cs) contributes 20-30%. Uncertainties in exposure parameters (radionuclide concentrations and amounts of exposure) are dominant for all pathways.

Results of Special Scenarios

Some individuals are thought to have consumed ground fish bones as well as flesh, in the form of fish patties. Therefore, an evaluation was made of the doses and risks resulting from substitution of part (8-20%) of a Category I consumer's fish intake with fish patties rather than flesh alone. The doses and risks to bone and red bone marrow are increased approximately 15-25% due to the increased ingestion of ^{90}Sr from the fish bones. However, because ^{90}Sr is a small contributor to total dose and risk from fish ingestion, the overall risk is not increased significantly by consumption of fish patties.

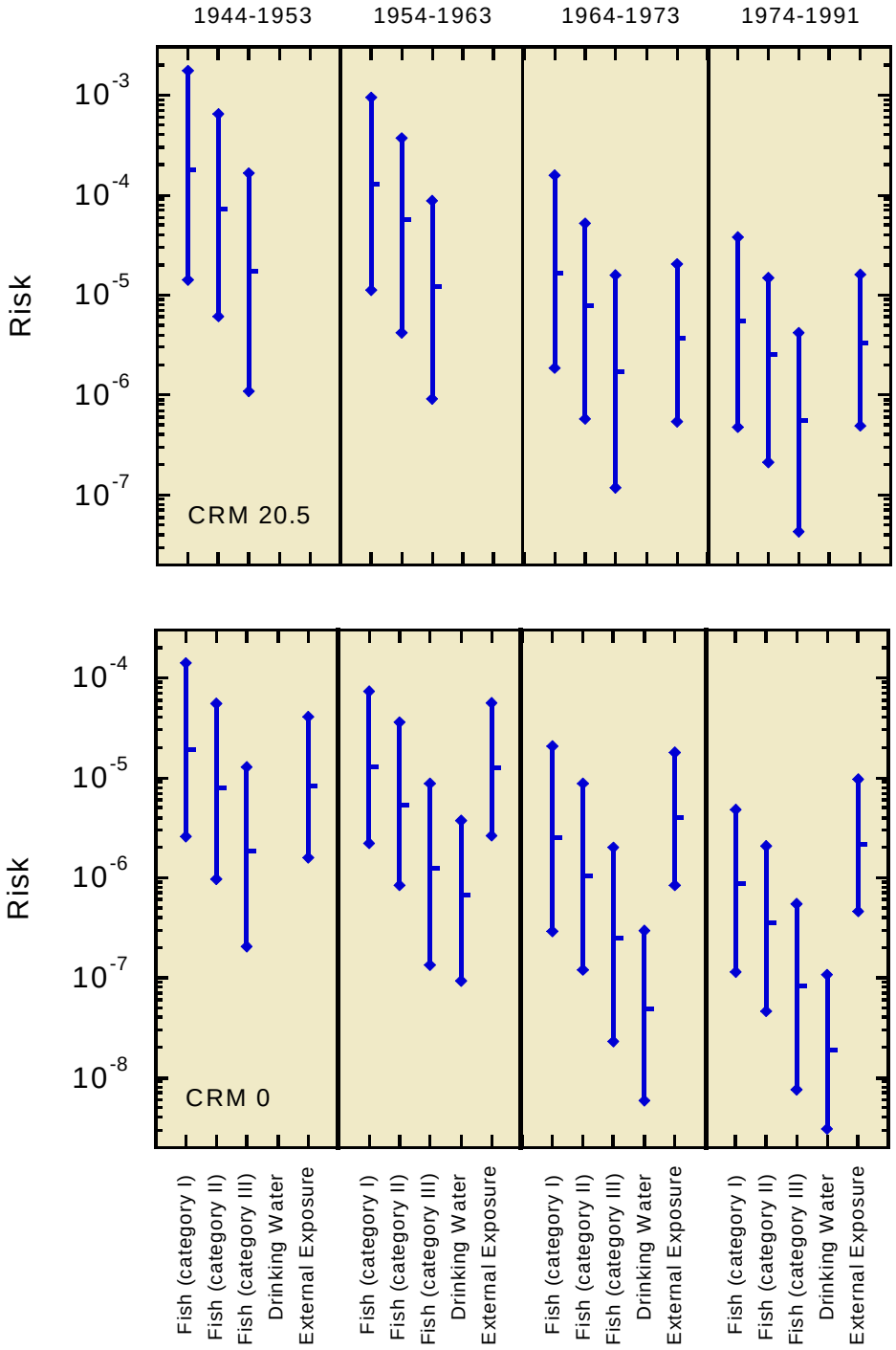

Figure ES.3 Excess lifetime risk of cancer incidence for males from fish ingestion, drinking water ingestion, and external exposure to shoreline sediment during four specific time periods for CRM 20.5 (top) and CRM 0 (bottom). The vertical lines indicate the 95% subjective confidence intervals on the risk estimates; the central values (50th percentiles) are also indicated. Risks for females from fish ingestion are slightly lower than those for males (see Fig. ES.1).

Four additional scenarios for internal exposure to radionuclides were evaluated, specifically the consumption of contaminated wildlife (fish, turtles, deer, or waterfowl) from the Oak Ridge Reservation. The risks per meal (4–16 ounces) were estimated for the highest reported contaminant levels and for more likely levels. For the most contaminated animals, the risk per meal ranged as high as 3×10^{-4}. Risks per meal for more likely values did not exceed 2×10^{-6}. For deer and waterfowl, risks were also calculated on a per animal basis. The number of people who may have been exposed to contaminated animals from the Oak Ridge Reservation cannot be determined precisely, but it is thought to be a very small fraction of the total population exposed to contaminated fish, water, or sediment.

Advancements in Dose Reconstruction Studies

This report highlights several advancements in the field of dose reconstruction:

(1) Environmental measurements of radionuclide concentrations in water and mathematical models for predicting radionuclide concentrations in water and sediment were combined in a single analysis.

(2) Both modeled and measured concentrations of radionuclides in water and sediment were adjusted for the existence of known sources of bias and uncertainty.

(3) Site-specific data in conjunction with an evaluation of the scientific literature were used to estimate site-specific bioconcentration factors for Clinch River fish.

(4) Detailed information about the demography of the region from 1944 to 1991 permitted the specification of categories of individuals who could have been exposed and thereby the characterization of the variability among individuals in the exposed population.

(5) Every effort was made to ensure a realistic analysis of exposure, dose, and risk, and all sources of uncertainty were included in the final risk estimates.

(6) This dose reconstruction is one of the first in which uncertainty in both external and internal dosimetry is expressed explicitly and the dose-response relationship of cancer incidence with its uncertainty is expressed for each organ and for total cancers.

(7) Extending the calculations to risk accounted for differing radiosensitivity among organs and permitted accurate identification of the organs of most importance.

Conclusions

The radiological doses and excess lifetime cancer risks estimated in this report are incremental increases above those resulting from exposure to background sources of radiation in the East Tennessee region. Nevertheless, for the exposure pathways considered in this task, the doses and risks are not large enough for a commensurate increase in health effects in the population to be detectable, even by the most thorough of epidemiological investigations. In most cases, the estimated organ doses are clearly below the limits of epidemiological detection (1 to 30 cSv) for radiation-induced health outcomes that have been observed following irradiation of large cohorts of individuals exposed either *in utero*, as children, or as adults.

Even in the case of Category I consumers of fish, the upper confidence limits on the highest estimated organ-specific doses are below 10 cSv, and the central values are below 1 cSv. The lower confidence limits on these doses are well below levels that have been considered as limits of epidemiological detection in studies of cohorts of other exposed populations. The large uncertainty, combined with the small number of individuals comprising Category I consumers, diminishes the statistical power available to detect a dose response through epidemiological investigation. Therefore, it is unlikely that any observed trends in the incidence of disease in populations that utilized the Clinch River and Lower Watts Bar Reservoir after 1944 could be conclusively attributed to exposure to radionuclides released from the X-10 site, even though this present dose reconstruction study has identified increased individual risks as high as 1×10^{-3} resulting from these exposures.

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1.0 INTRODUCTION

From 1944 to the present time, the Oak Ridge National Laboratory has discharged liquid radioactive wastes into White Oak Creek, which empties into the Clinch River at mile 20.8 (CRM 20.8). Initially, the Oak Ridge National Laboratory (ORNL; originally called the Clinton Laboratory) was designed to operate for one year with all the resulting radioactive waste to be stored in a series of gunite tanks. However, the scope of the work at ORNL was rapidly expanded, and as a result, more radioactive waste was produced. A decision was made in early 1944 to release low-level radioactive wastes to the environment via White Oak Creek. Over time, the radioactive wastes came from the direct operations of the graphite reactor and the chemical separation plant as well as other ORNL operations and other nuclear facilities throughout the southern region. Significant off-site releases of radioactive wastes continued until the mid-1960s, when dramatic reductions in releases occurred because of changes in ORNL operations and modifications to ORNL radioactive waste disposal. (See Section 2 of this report for a detailed discussion of radioactive waste disposal from the Oak Ridge National Laboratory.)

Potential exposures to individual members of the public from radionuclide releases have existed since the first releases of radioactivity to the river in 1944. The public has had access to the Clinch River, but not to White Oak Creek, which is located on the Oak Ridge Reservation, where access is restricted and controlled. The purpose of Task 4 is to calculate the total lifetime risk of cancer for specific target individuals exposed to radioactivity resulting from releases of radionuclides to the Clinch River from 1944 to 1991.

To obtain information for calculating the doses and risks, the following Task 4 objectives were established:

(1) To conduct a screening analysis of the radionuclides released into the Clinch River and of all potential exposure pathways, in order to determine whether some of the radionuclides or exposure pathways could be considered low priority for further study;
(2) To determine the quantity of radionuclides released from White Oak Dam each year from 1944 to 1991 (source term);
(3) To determine the concentration of radionuclides in the water and sediment at downstream locations in the Clinch River from 1944 to 1991 by evaluation of available measurements and by modeling the radionuclide releases to the Clinch River from White Oak Dam;
(4) To conduct a demographic study of the human population that lived along the Clinch River from 1944 to 1991 to determine the activities and dietary habits of critical individuals and the potential for these individuals to be exposed to radionuclides in one of the Clinch River pathways;
(5) To identify locations and exposure pathways for calculating radiation doses and risks to specified target individuals;
(6) To determine the concentration of critical radionuclides in Clinch River fish by evaluation of site-specific bioconcentration factors (ratio of the concentration of radionuclide in fish to the concentration of radionuclide in water);

(7) To determine the uncertainty of the dose factors for critical organs for the most important radionuclides;
(8) To determine the doses to specific organs and to the whole body, together with the subsequent health risks, including uncertainty in these estimates.

The screening analysis (Section 3) identified radionuclides and exposure pathways for which the human health risk is clearly below a level of concern (1×10^{-5})[1] so that Task 4 could focus on evaluating radionuclides and pathways that were more likely to have been important in terms of doses and risks to individuals who resided on or utilized the Clinch River downstream from the Oak Ridge National Laboratory facilities. Twenty-four radionuclides released into the Clinch River from 1944 -1991 were included in the screening analysis. Three external exposure pathways were considered: swimming, exposure to shoreline sediment, and exposure to dredged sediment. Internal exposure pathways were considered for radionuclides ingested in contaminated water, fish, meat, milk and agricultural products. The screening analysis was performed for a hypothetical individual receptor who engaged in fishing and swimming and was involved in agricultural activities at a site on Jones Island. As a result of the screening analysis, eight radionuclides (^{137}Cs, ^{90}Sr, ^{106}Ru, ^{60}Co, ^{131}I, ^{144}Ce, ^{95}Nb, and ^{95}Zr) were carried into the next round of study, while sixteen radionuclides were assigned a low priority for further analysis. Of the eight remaining radionuclides, ^{137}Cs, ^{60}Co, ^{106}Ru, and ^{90}Sr were expected to be the most important. In addition, the swimming and irrigation pathways were assigned low priority because of low estimates of exposure and the fact that the only documented incidence of the use of river water to irrigate crops was for a small acreage of peaches.

To calculate realistic radiation doses and risk estimates to populations exposed to releases of radioactivity to the Clinch River, the types of individuals who used the river and the pathways by which they were exposed were first identified. A demographic study was conducted for the area from Highway 95 downstream to the City of Kingston, which is located at the confluence of the Clinch and Tennessee Rivers. Although many sources of information were investigated, interviews with individuals who had lived along the Clinch River and surrounding areas from 1944 to 1991 proved to be the most detailed. Information was obtained on the size and daily activities of the families, on the dietary habits of the families, and on their use of the river for recreational and agricultural practices. The demographic study (Section 7) helped to establish the location of the critical individuals and to identify the most important exposure pathways in each of five sections of the Clinch River from CRM 20.8 to CRM 0.0.

One of the critical pathways of exposure for individuals living along the Clinch River is the ingestion of contaminated fish, because fishing was and is a major recreational pastime. Measured concentrations of radionuclides in fish in the Clinch River are available for some years, locations, and species of fish. However, for most of the years when radionuclide concentrations in the water were greatest, measured concentrations of radionuclides in fish are not available. Therefore, values known as "bioconcentration

[1] A value of 1×10^{-4} disease incidence was selected by the Oak Ridge Health Agreement Steering Panel as a decision guide for screening assessments. Because the risks were not added across pathways in the screening analysis, a lower risk level of 10^{-5} was used as the decision guide.

factors"[2] were used to estimate the concentrations of specific radionuclides in fish as a function of the radionuclide concentration in the river water. Bioconcentration factors specific for fish in the Clinch River were developed for ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co from available measurements of radionuclide concentrations in Clinch River fish and water, together with published information on the behavior of radionuclides in rivers and in fish and with information on local water conditions (Section 8).

Calculation of the radiation doses to exposed individuals in the present study also involves consideration of the transfer of selected radionuclides (e.g., ^{131}I) from river water to milk and beef (Section 9), uncertainties in external dose factors in the shoreline exposure pathway (Section 10), and uncertainties in the internal dosimetry of specific radionuclides in specific organs (Section 11). For internal dosimetry the uncertainty associated with the International Commission on Radiological Protection (ICRP) organ-specific internal dose factors was determined. Biokinetic information for each of the radionuclides was investigated, and the parameters responsible for the uncertainty were determined for the organ-specific internal dose conversion factors for ^{137}Cs, ^{60}Co, ^{106}Ru, ^{90}Sr, and ^{131}I (Section 11). Uncertainties associated with the external dose factors for ^{137}Cs, ^{60}Co, ^{106}Ru, ^{90}Sr, ^{144}Ce, ^{95}Nb, and ^{95}Zr were also determined (Section 10). Lifetime risk calculations were based on the dose-response relationships obtained from the epidemiological study of the Japanese Atomic Bomb survivors. The dose-response relationships were transferred to U.S. populations and adjusted for all known potential sources of error. The risk estimates in this study are based on cancer incidence rather than mortality for most organs or cancer sites of interest.

The following Task 4 report contains the calculated radiation doses and risk values, along with uncertainty estimates of the risk to potentially exposed target individuals (Section 13). These reference individuals are members of potentially exposed populations which could have been exposed by one or more pathways to radioactivity in the Clinch River from 1944 to 1991. The calculated risks to individuals exposed to radionuclides released to the Clinch River are evaluated in terms of the increased risks of cancer incidence.

[2]By definition, a bioconcentration factor is the ratio of the radionuclide concentration in fish to the concentration in water when at equilibrium in the environment. Many factors such as water quality and the eutrophic state of the system can influence the value of the concentration factors.

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2.0 HISTORY OF RADIOACTIVE WASTE DISPOSAL AT THE X-10 SITE

The main facilities of the X-10 site, now the Oak Ridge National Laboratory (ORNL), are situated in Bethel Valley, which runs approximately northeast-southwest. The terrain surrounding ORNL is mostly wooded. White Oak Creek, a small stream originating in Bethel Valley, flows southward adjacent to the Laboratory and enters Melton Valley through a gap in Haw Ridge. From Melton Valley, White Oak Creek enters the Clinch River at Mile 20.8[1] (Figures 2.1 and 2.2).

Construction of the Oak Ridge National Laboratory, originally called the Clinton Laboratory, began on February 10, 1943. The Laboratory was built as a pilot plant for demonstrating the production and separation of plutonium. Major construction projects at the time were the Graphite Reactor, the "Hot Pilot Plant" (a chemical separation plant), and a number of large underground concrete "Gunite" tanks to store highly radioactive waste. Originally the Graphite Reactor was to operate for one year, with all highly radioactive waste to be stored in the Gunite tanks.

The Graphite Reactor went into operation on November 11, 1943. From December 1943 to January 1945, the primary objective of the Laboratory was to produce gram quantities of plutonium per day. As a result, large quantities of radioactive waste were produced. During this same time period, the mission of ORNL was expanded from one year to a state of indefinite continuation. The scope of work was broadened to include additional fuel processing research, large-scale production of radioisotopes, and operation of several other experimental reactors in addition to the Graphite Reactor. The additional work necessitated a method to augment the storage of radioactive waste being held in the Gunite tanks. The management of large quantities of radioactive waste was a unique problem in the early 1940s. The expanding mission of ORNL meant that the wastes had to be dealt with expeditiously, even though methodologies for dealing with radioactive wastes were still in an experimental stage.

The Gunite tanks were geographically divided into the North Tank Farm and the South Tank Farm (Browder, 1949). The tanks were buried at least 6 feet deep. The function of the tank farms was to receive and store radioactive metal waste and to collect and hold radiochemical waste until sufficient radioactivity was lost either by physical decay or through precipitation of certain radionuclides so that the liquid effluent could be discharged.

In 1943 White Oak Creek was impounded 0.6 miles upstream from its mouth to form White Oak Lake. Releases of radioactive waste from ORNL could be held in White Oak Lake for a few days to permit suspended solids to settle and to allow time for short-lived radionuclides to decay.

Sampling methods at White Oak Dam varied from intermittent grab samples in the beginning to more sophisticated flow-proportional monitors in later years. Until 1990, White Oak Creek flowed the final 0.6 miles unimpeded in what is known as the White Oak Creek Embayment before joining the Clinch River at Mile 20.8. In 1990, a coffer cell dam was constructed at the mouth of the creek to prevent radioactive sediment that had been deposited in the embayment from eroding into the Clinch River (Blaylock et al., 1993).

[1] Rivers and creeks are typically measured in terms of miles or kilometers upstream from their mouths, the points where they empty into other bodies of water. For example, Clinch River Mile 20.8 (CRM 20.8) is 20.8 miles upstream from the point where the Clinch River joins the Tennessee River near Kingston, TN.

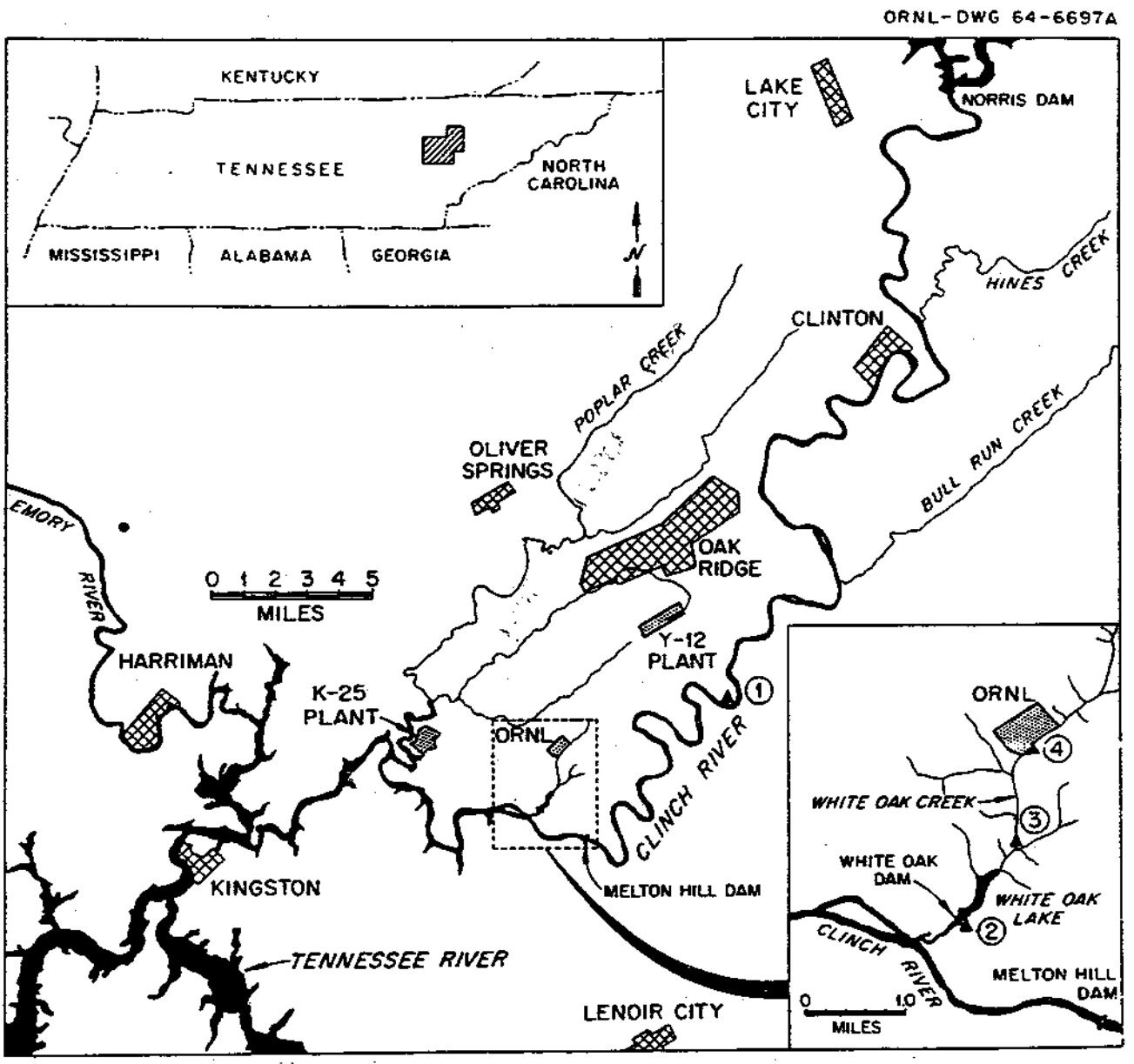


Figure 2.1 Location of the Oak Ridge Reservation in East Tennessee and the associated lakes and rivers (Churchill et al., 1965).

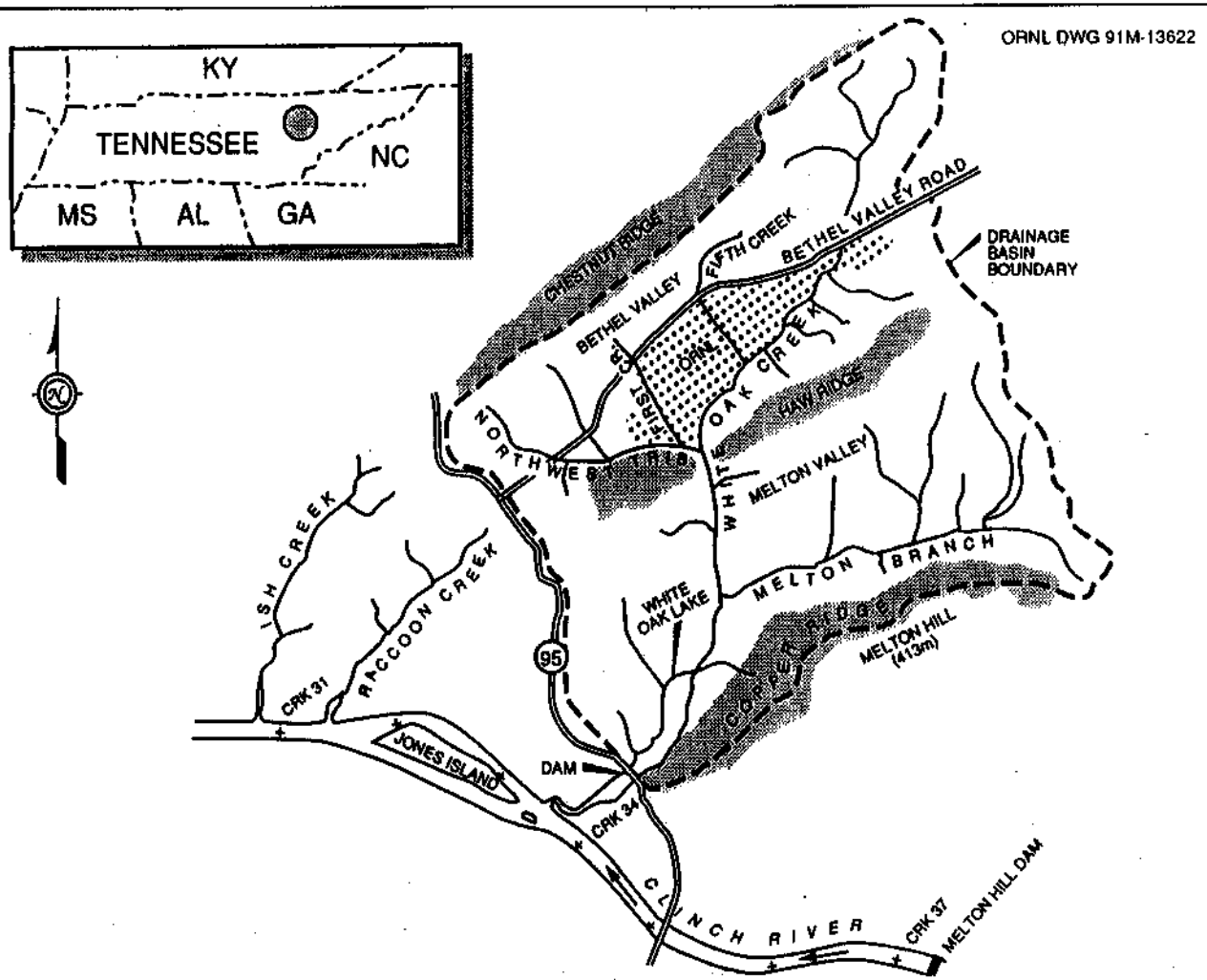


Figure 2.2 Location of the Oak Ridge National Laboratory in relation to White Oak Creek, Melton Branch and White Oak Dam. White Oak Creek empties into the Clinch River at Mile 20.8 about 2.5 miles below Melton Hill Dam, which was impounded in 1963.

2.1 Waste Disposal 1944-1949

From 1944 to 1949, "highly" radioactive waste was placed in the underground Gunite storage tanks. The fluids that did not settle (supernatants) were decanted and released to White Oak Creek via the Settling Basin (Figure 2.3) after being mixed with process water from ORNL. Originally, the process water containing radioactive waste went into one of two holding ponds, the East and West Ponds (Figure 2.3), which had a capacity of about 293,200 gallons (1100 m^3) each. The ponds were to alternately receive releases of the supernatant and process water from Gunite tanks W-5 and W-6 in the North Tank Farm. The water in the East Pond was to "cool" through the decay of short-lived isotopes and the settling out of "hot" particles while the West Pond was being filled. The East Pond would be drained into White Oak Creek when the West Pond was nearly full and vice versa (Browder, 1959).

Radioactive waste discharge into the East and West Ponds began March 6, 1944, and was discontinued April 27, 1944, when Overstreet and Jacobson reported relatively high levels (up to 37,000 Bq g^{-1} or 1×10^6 pCi g^{-1}) of fission products in the water and mud of White Oak Creek and in the Clinch River near the mouth of White Oak Creek (Overstreet and Jacobson, 1944; Morgan and Western, 1947). The inadequacy of the East and West Ponds to contain the radioactivity was largely due to the small size of the ponds and to the fact that much of the precipitate that settled to the bottoms of the ponds washed into White Oak Creek when the ponds were drained.

Because of the inadequacy of the holding ponds, a 1,500,000-gallon (5700-m^3) Settling Basin was built and put into operation on July 3, 1944. The discharge from Gunite tanks W-5 and W-6 was mixed with the process water from the Laboratory to give a flow of about 900,000 gallons (3400 m^3) of water per day into the Settling Basin. The Settling Basin was to serve as a collection, sampling, and settling pond for the Gunite tank supernatants to permit radioactive solids to settle from the waste water before it was discharged to White Oak Creek (Feige et al., 1960). Radioactive particulates in the water had more time to settle out in the Settling Basin than in the smaller retention ponds, and the activity released to White Oak Creek was greatly reduced. A limit of 5 Ci (1.9×10^{11} Bq) of radioactivity per day was established for the discharge into White Oak Creek (Browder, 1949). This limit was in effect from 1946 to 1948. The Settling Basin operated from 1944 to 1976. In 1985, the estimated inventories of radionuclides in the Settling Basin (currently Pond 3513) were as follows: 130 Ci (4.8×10^{12} Bq) of ^{137}Cs; 1 Ci (3.7×10^{10} Bq) of ^{60}Co; 20 Ci (7.4×10^{11} Bq) of ^{90}Sr; 0.1 Ci (3.7×10^{9} Bq) of ^{238}Pu; 3 Ci (1.1×10^{11} Bq) of ^{239}Pu; 0.5 Ci (1.9×10^{10} Bq) of ^{241}Am; 0.1 Ci (3.7×10^{9} Bq) of ^{244}Cm; and 0.2 Ci (7.4×10^{9} Bq) of ^{154}Eu (Stansfield and Francis, 1985).

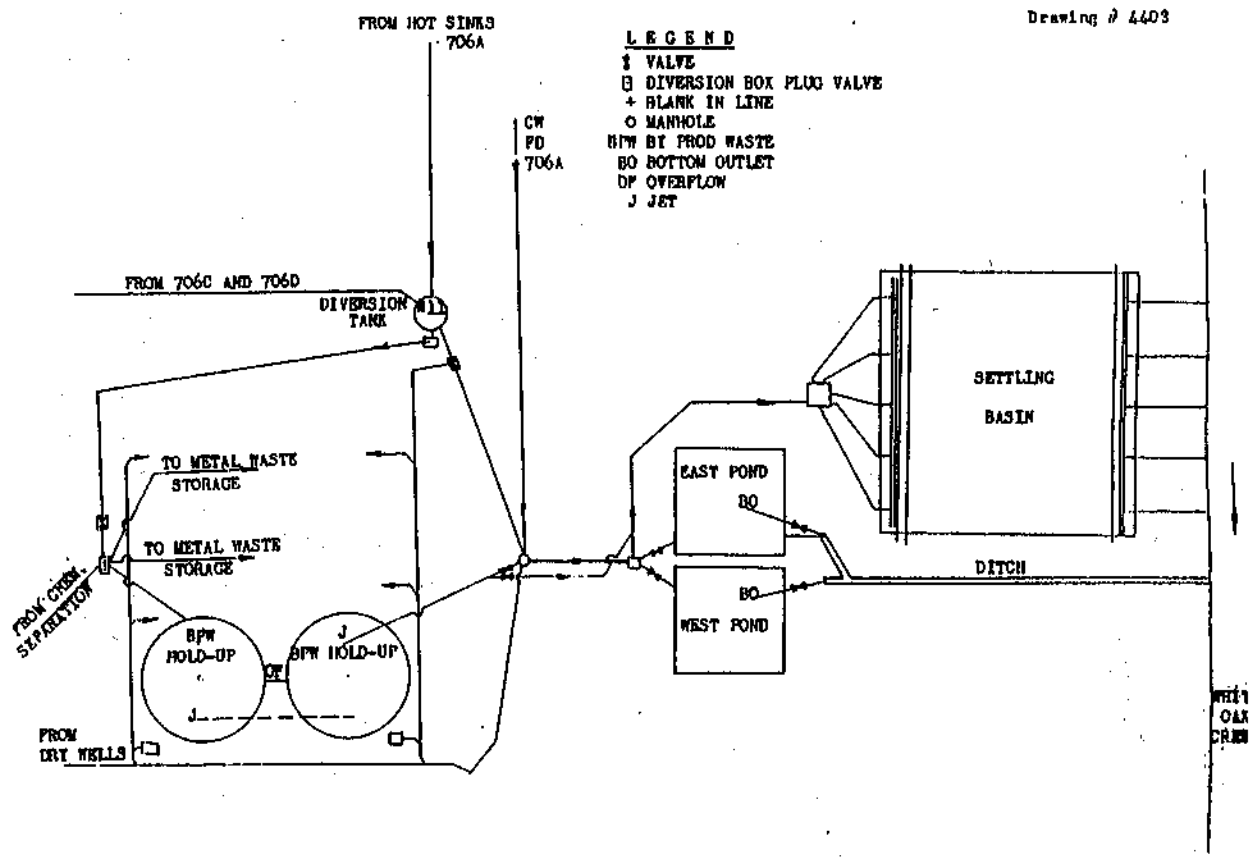


Figure 2.3 A drawing of the East and West Ponds and the Settling Basin in 1944. Radioactive effluents were released into either the East or West Pond; one pond was then emptied into White Oak Creek while the other was filled. Releases to the ponds began March 6, 1944, and ceased on April 26, 1944, after high levels of radioactivity were discovered in White Oak Creek and the Clinch River. A larger settling basin (3513) was then constructed and went into operation on July 3, 1944, receiving decanted waste from the Gunite tanks and laboratory waste water (After Morgan and Western, 1947).

2.2 Waste Disposal 1949-1954

Identification of different categories of liquid wastes was an important part of the disposal process. In 1949, liquid waste was classified into four types according to composition and radioactivity (Browder, 1949).

Radiochemical Waste was classified as "highly" radioactive (averaging 2.5×10^5 beta counts/minute/milliliter) and had as its source special "hot" sinks and vessels in cells devoted to "hot" work. In 1949, the average volume was 30,000 gallons (110 m^3) per week. [Note: This "highly" radioactive waste was later defined as intermediate-level waste (ILW)(Coobs and Gissel, 1986).]

Metal Waste included plutonium-, uranium-, or thorium-bearing waste which was usually highly radioactive (averaging 10^5 beta counts/minute/milliliter). In 1949, the metal waste averaged 1,500 gallons (5.7 m^3) per week.

Warm Waste was moderately radioactive (averaging 10^4 beta counts/minute/milliliter). In 1949, warm waste averaged 75,000 gallons (280 m^3) per week.

Process Waste was theoretically nonradioactive and was derived from cooling water, laboratory sinks other than "hot" sinks, and floor drains. However, the Graphite Reactor canal overflow also drained into this system. The process waste volume was two to four million gallons (7,600-15,000 m^3) per week and averaged 100 beta counts/minute/ milliliter.

In 1949, the precipitation and decanting procedure for treatment of "highly" radioactive waste was replaced by an evaporation step that produced both a concentrate, which was permanently stored in the Gunite tanks, and a low-level condensate, which was discharged into White Oak Creek. A pot-type waste evaporator along with necessary auxiliary equipment was placed in operation at ORNL in June 1949 (Straub, 1956). From June 1949 until June 1954, all "highly" radioactive liquid chemical waste was concentrated by evaporation instead of by the procedure involving decanting, supernatant dilution, and release into White Oak Creek. During this period, the evaporator processed a total of 11,650,000 gallons (44,000 m^3) of waste and reduced the volume to 432,000 gallons (1600 m^3) of radioactive concentrate, which was stored in the Gunite tanks. Other radioactivity entering White Oak Creek came from process waste water and from accidental discharges. The evaporator was taken out of service in June 1954 after a 1,000,000-gallon (3800-m^3) experimental in-ground, waste disposal pit had been in use for about two years.

2.3 Waste Disposal Pits and Trenches 1951-1970

Seven open pits and covered trenches were built and operated between 1951 and 1966 for the disposal of radioactive liquid waste. The pits (1-4) and trenches (5-7), located in the White Oak Lake drainage area (Figure 2.4 and Photograph 1), were intended to dispose of radioactive waste by letting the liquid seep from the pits and trenches while the radioactivity was retained by the surrounding soil. About 1.2 million curies (4.4×10^{16} Bq) of high-energy beta emitters were disposed of using this method (Spalding and Boegly, 1985). Until 1959, open seepage pits were used exclusively for the disposal of all "highly" radioactive liquid waste (later classified as ILW), and, at the time, the operation was considered safe, simple, and economical.

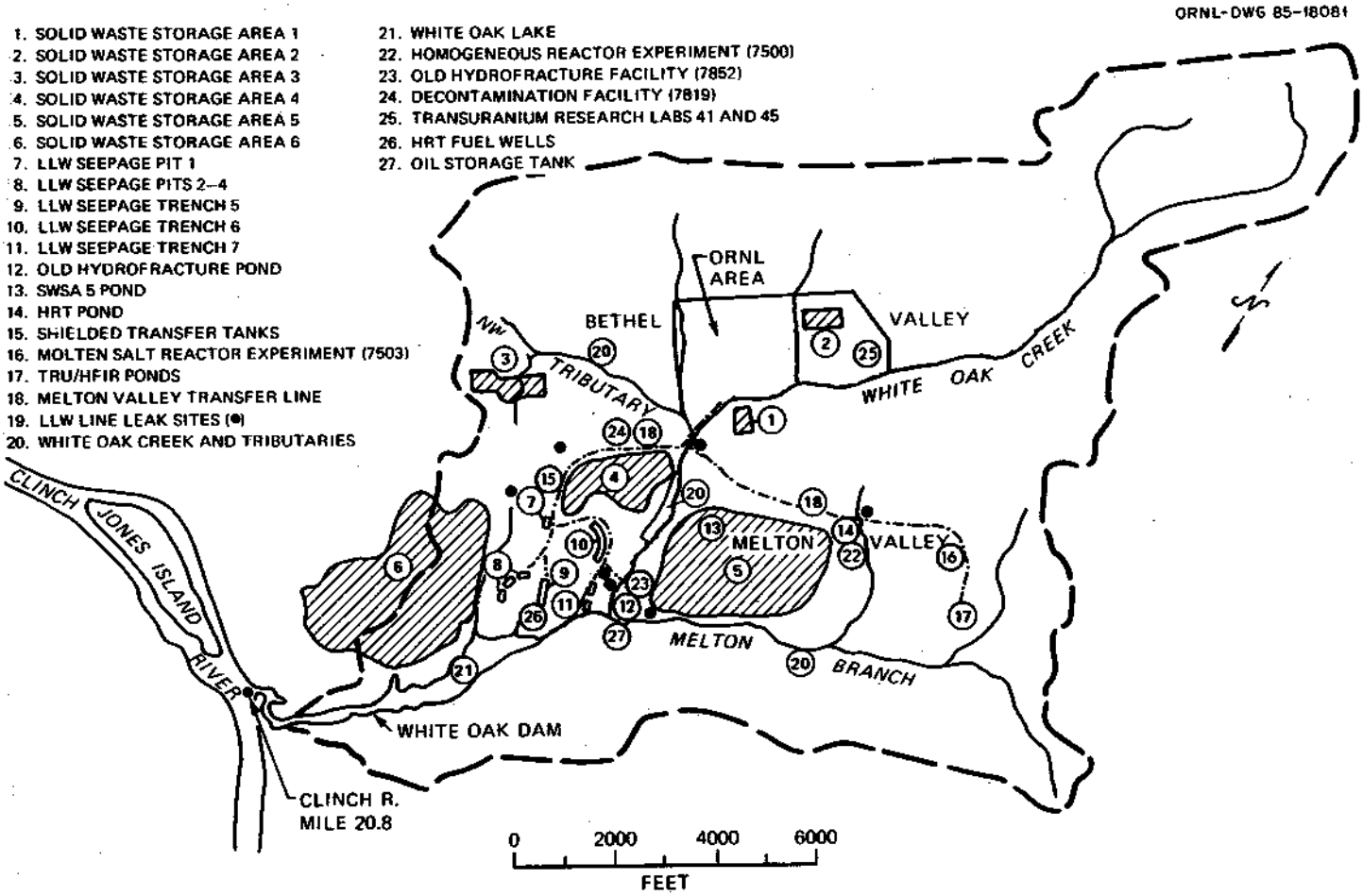
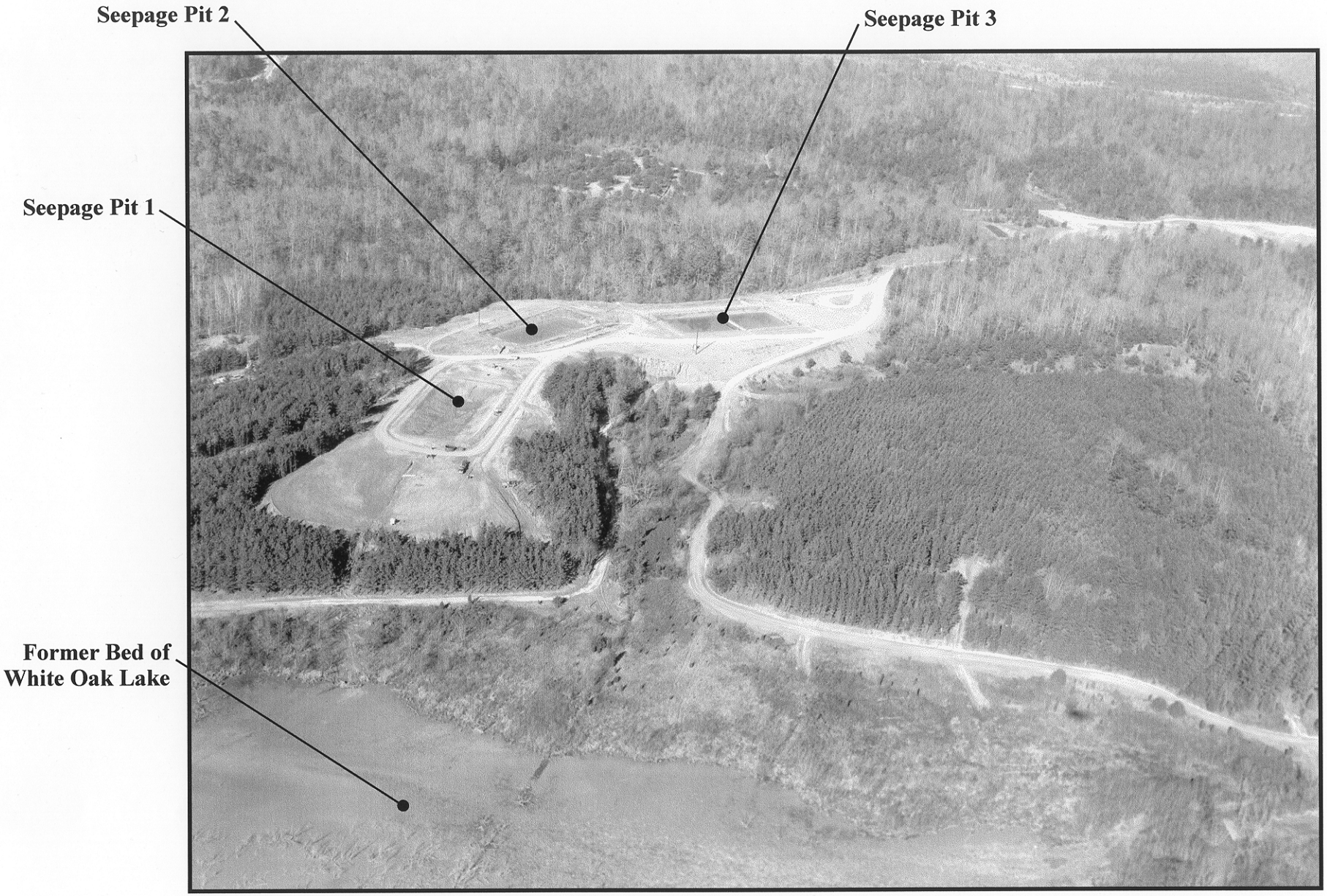


Figure 2.4 Locations of waste (seepage) pits 1-4 and trenches 5-7 in relation to White Oak Creek, approximately 0.5 miles (800 m) above White Oak Dam.



Photograph 1: Seepage pits 2,3 and 4 and the location of the seepage trenches in relation to the former White Oak Lake bed (White Oak Lake was drained at the time). View is toward the east.
(Photo 15895 courtesy of Oak Ridge National Laboratory.)

Photograph 2: Waste pit 2 and a tanker truck that was used to transport liquid radioactive waste to the pit before construction of the pipeline from the Gunite tank to the seepage pit area (1954).
(Photo 12116 courtesy of Oak Ridge National Laboratory)

Photograph 3: A view of White Oak Lake bed shortly after draining in 1955, looking southward toward Highway 95, which crosses White Oak Dam. The meandering stream below the dam flows through White Oak Creek embayment to the Clinch River in the background; the Clinch is flowing to the right.
(Photo 15093 courtesy of Oak Ridge National Laboratory)

Strontium retention in the pits was good, and ^{90}Sr was considered the controlling contaminant in establishing the MPC (Maximum Permissible Concentration) for the Clinch River. It was discovered, however, that ruthenium, unlike strontium, was not retained by the soil. Because of ^{106}Ru's short half-life (about 1 year), the importance of these releases was not recognized until 1959, when large quantities of ^{106}Ru were reported in groundwater leaving the pits. This followed a sharp increase in radioactivity discharged into the pits in 1959 that was due mainly to higher levels of radioactivity in waste from the Hot Pilot Plant, where chemical separation of radionuclides was performed. A potentially serious release of ^{106}Ru from the east bank of Waste Pit 4, occurring in late 1959, caused the release of an estimated 1320 curies (4.8×10^{13} Bq) to White Oak Creek. At the same time, the concentration of ^{106}Ru in the Clinch River began to increase, and radiation levels at the pit site became too high to send personnel into the area for sampling or other purposes.

2.3.1 Waste Pits

Waste Pit 1, was constructed in July 1951 with a capacity of 180,000 gallons (680 m^3). About 123,000 gallons (470 m^3) reportedly containing 389 curies (1.4×10^{13} Bq) of gross beta radioactivity were hauled from the evaporator to Pit 1 in 500-gallon (2-m^3) tanks fitted on a Dempster Dumpster truck. With the discovery of radioactive leakage from the pit, discharges into the pit were terminated October 5, 1951. Leakage was attributed to the poor location of the pit relative to the underlying terrain. The waste was highly alkaline and contained about 60% ^{137}Cs and 40% ^{106}Ru, but little ^{90}Sr.

Waste Pit 2, completed June 1952 and situated on a hillock southwest of Pit 1, had a capacity of about 1,000,000 gallons (3800 m^3) (Photograph 2). In 1954, a pipeline was constructed from the Gunite tanks to Pit 2, and the waste evaporator was shut down. Typically 3,000 to 7,000 gallons (11-26 m^3) per day were pumped to Pit 2. Through December 1954, when Pit 2 was operating alone, 1,294,000 gallons (5000 m^3) of liquid waste containing a reported 16,600 curies (6.1×10^{14} Bq) of beta activity were pumped to the pit. Pit 2 probably received no waste in 1955, because Pit 3 was opened in January and took almost a year to fill. Pit 2, along with Pits 3 and 4, received some sludge hauled by tank trailer from the Process Waste Treatment Plant after it began operating in 1957. In October 1959, a potentially serious breakthrough of ^{106}Ru activity occurred in the bank on the east side of Pit 2. During this period, an estimated 300 to 350 curies ($1.1\text{-}1.3 \times 10^{13}$ Bq) of activity, primarily ^{106}Ru, were released into White Oak Creek from Pit 2. Pit 2 was covered with asphalt in 1970.

Waste Pit 3, with a capacity of 1,000,000 gallons (3800 m^3), was completed and put into service in January 1955. Discharge of the liquid waste from the pipeline was to Pit 3, and when it became full, the overflow was piped downhill to Pit 2. The quantity of waste placed in Pit 3 is difficult to estimate. Lomenick et al. (1967) estimated that Pits 2 and 3 contained essentially all of the ^{90}Sr and about 85% of the ^{137}Cs discharged to the pits. Lesser amounts of ^{89}Sr, ^{90}Sr, ^{60}Co, and ^{125}Sb were present. Neither ^{3}H nor ^{99}Tc was measured because they are low-energy beta emitters (Cowser et al., 1961; Lomenick et al., 1967). In addition, seepage rates from the pits were difficult to calculate. Groundwater seeps were observed on the east side of Pit 3, but were not as severe as those of Pit 2 or later seeps found originating from Pit 4. Pit 3 was taken out of service in September 1961, and backfilling and grading were completed in 1963.

Waste Pit 4, was completed in October 1955, but was not put into service until April 1956, when it first received overflow of liquid waste from Pit 2. This seepage pit was similar in design to Pits 2 and 3 and was located downhill from Pit 2. By the time Pit 4 went into operation, the Laboratory depended heavily on the pits for disposal of liquid waste. Pit 4 started leaking almost immediately and never held liquid to the degree of Pits 2 or 3. Because it was at the end of the overflow from the other pits, it did not receive a large volume of waste until 1959, when some large discharges of ^{106}Ru entered the pits. Leakage

occurred on the east side of Pit 4, and as a result, an interceptor trench 10 feet deep and 175 feet long was dug on the east side. For some time during 1959, approximately 50 Ci (2×10^{12} Bq) of ^{106}Ru per day in 8,000 gallons (30 m^3) of waste were pumped back into Pit 4. Although the discharge of elevated levels of ^{106}Ru to the pits was curtailed, seepage of ^{106}Ru from the pit area remained high for several years. When a new process waste treatment plant became operational in 1976, Pit 4 was gradually backfilled and eventually paved with asphalt in 1980.

Prior to November 1962, Pits 2, 3, and 4 received a total inventory of 522,565 curies (1.93×10^{16} Bq) of gross beta activity. Lomenick et al. (1967) estimated that this total represented 42,000 Ci (1.5×10^{15} Bq) of ^{90}Sr; 184,000 Ci (6.8×10^{15} Bq) of ^{137}Cs; 230,000 Ci (8.5×10^{15} Bq) of ^{106}Ru; and about 70,000 Ci (2.6×10^{15} Bq) of trivalent rare earths (such as Y, Sc, Ce, Pr, Eu).

2.3.2 Waste Disposal Trenches

In response to the increasing demand for liquid waste disposal space at ORNL, waste disposal Trench 5 was constructed in May 1960. Trenches were numbered in sequence after the waste pits; i.e., there were no trenches 1-4. Changes were incorporated in the design of these seepage trenches (Figure 2.5) to overcome the following operational problems encountered with the pits:

- intense radiation fields developed around the three seepage pits;
- netting was required over the open pits to prevent use by waterfowl;
- direct precipitation entered the pits;
- seepage rates were difficult to measure in the open pits; and
- seepage occurred along the walls of the pits.

Trench 5, completed May 20, 1960, was 300 ft long by 15 ft deep (91 m × 4.6 m). The trench was treated with 1,800 pounds (820 kg) of copper sulfate and 1,000 pounds (450 kg) of sodium sulfide in an attempt to overcome the ^{106}Ru mobility problem. Then the trench was filled with coarse rock and covered with soil to reduce the external exposure. Liquid waste generated at the Laboratory averaged 7,000 to 8,000 gallons (26-30 m^3) per day, necessitating the continued use of the pits as well as Trench 5. Trench 5 was probably the most effective trench, built with no specific areas of drainage. Trench 5 operated at about maximum capacity until 1966, after receiving 9.5 million gallons (36,000 m^3) of waste containing 311,824 gross beta curies (1.15×10^{16} Bq). The estimated radionuclide contents of this waste were as follows: 96,750 Ci (3.6×10^{15} Bq) of $^{89\text{-}90}$Sr; 205,600 Ci (7.6×10^{15} Bq) of ^{137}Cs; 6,385 Ci (2.4×10^{14} Bq) of ^{106}Ru; and 3,045 Ci (1.1×10^{14} Bq) of ^{60}Co. The mounded earthen surface of Trench 5 was paved with asphalt in 1970.

The excavation and filling of the second waste trench, Trench 6, was completed in June 1961. This trench, which was U-shaped and 500 ft (150 m) in length, was about 50% larger than Trench 5. The size of Trench 6 should have accommodated all of the Laboratory's liquid waste; however, it was not located using geological precoring and water table information. Instead the location was apparently chosen because of its close proximity to the end of the liquid waste transfer line (Kertesz, 1961). Trench 6 received its first waste on September 7, 1961, and was taken out of service on October 10, 1961, when significant seepage of 0.15 gallons (5.7×10^{-4} m^3) per minute containing ^{90}Sr and ^{137}Cs was discovered; the radiation field was 20 mr/hr. Trench 6 received only about 130,000 gallons (490 m^3) of waste containing 145 Ci (5.4×10^{12} Bq) of ^{90}Sr, 665 Ci (2.5×10^{13} Bq) of ^{137}Cs, 501 Ci (1.9×10^{13} Bq) of ^{106}Ru, and 24 Ci (8.9×10^{11} Bq) of ^{60}Co. Trench 6 was covered with asphalt in 1981.

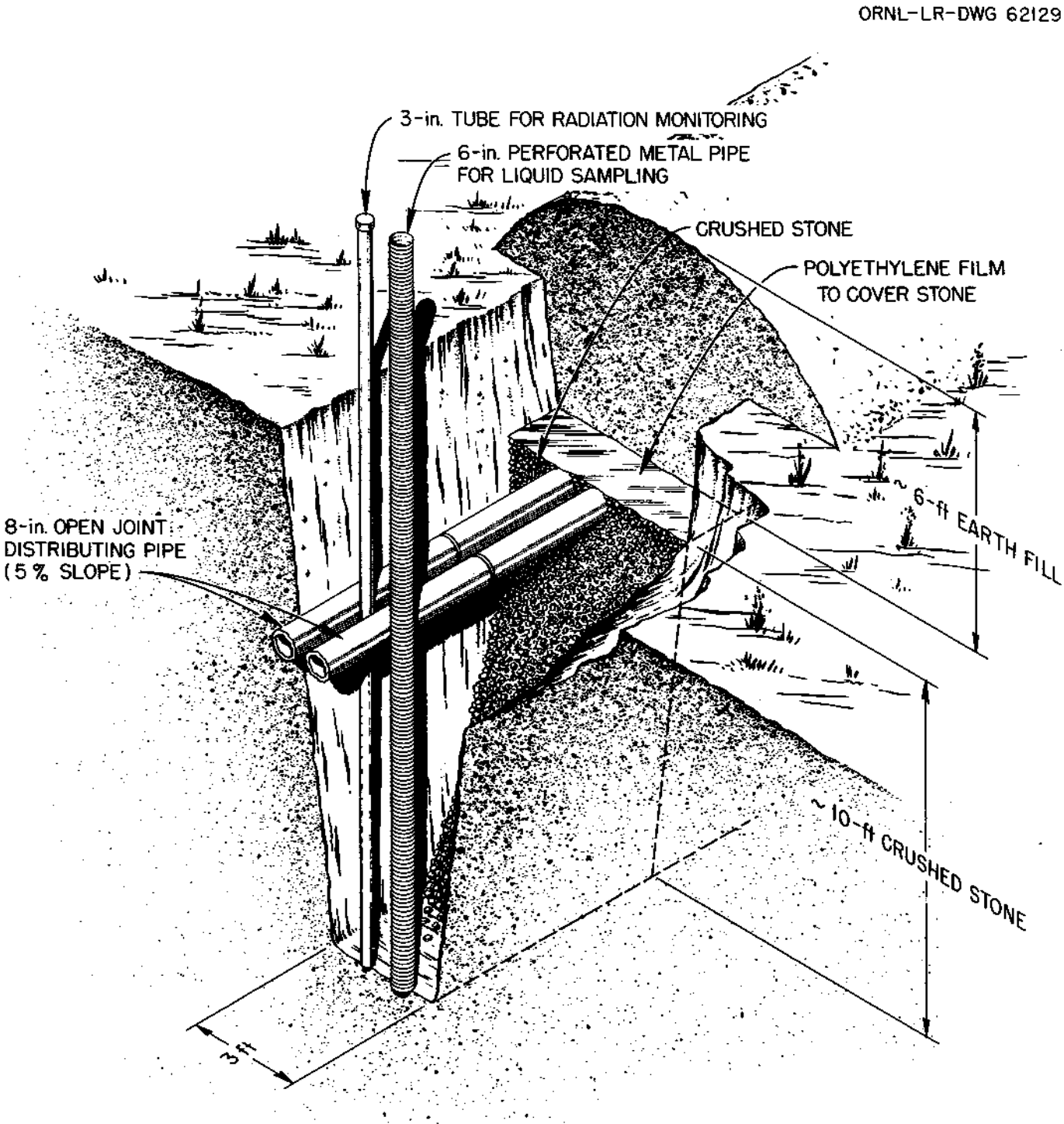


Figure 2.5 A drawing showing the improved design of the waste disposal trenches that succeeded the waste disposal pits. The main concern was to reduce the external exposure to workers below that encountered at the open pits.

The location and orientation of Trench 7, the last seepage trench, was based on geologic advice with preconstruction water table monitoring and geological coring to prevent the problems encountered with Trench 6. Trench 7 was designed to have three 100-ft (30-m) long segments that could be operated independently, so that if a leak occurred in one segment, the entire trench would not be lost.

Construction of Trench 7 was completed in August 1962. The third segment of Trench 7 was not completed because of high ground water elevations. The trench was treated with 50,000 gallons 190 m^3) of 4% sodium hydroxide to enhance the adsorption of ^{90}Sr. Trench 7 had about four times the seepage capacity of Trench 5, an amount that was greater than the liquid waste generation rate of the Laboratory. The capacity of Trench 7 allowed the two remaining pits, Pits 2 and 4, to be removed from service. Trench 7 performed well, handling the 9.5 million gallons (36,000 m^3) of waste transferred to it. This waste contained about 48,000 Ci (1.8×10^{15} Bq) of ^{90}Sr; 231,000 Ci (8.5×10^{15} Bq) of ^{137}Cs; 3,400 Ci (1.3×10^{14} Bq) of ^{106}Ru; and 1,500 Ci (5.6×10^{13} Bq) of ^{60}Co. When the hydrofracture facilities became operational in 1966, Trenches 5 and 7 were used for the last time (National Academy of Sciences, 1954). Trench 7 was paved with asphalt in 1970 when Pit 2 and Trench 5 were covered.

2.4 Process Waste Water Treatment Plant

A process water treatment plant was completed and put into operation August 8, 1957. The function of this plant was to reduce the level of radioactive contamination in the low-activity process waste water being discharged to White Oak Creek. An automatic diversion valve was installed in January 1959 to divert the entire process waste water flow to the treatment plant when the level of radioactivity in the water exceeded a given level. When the activity was below the set level, the valve diverted the water around the treatment plant and into the creek via the Settling Basin. Sludge waste from the process waste water treatment plant was taken to the pits and trenches for disposal. A replacement waste water treatment plant began operation in 1976.

2.5 White Oak Lake

2.5.1 Overview of the History of White Oak Lake

White Oak Lake has served as the final settling basin for radioactive liquid waste from the Laboratory since 1943. In 1941, the Tennessee Valley Authority placed a culvert and an earthen fill, 0.6 miles (1 km) upstream on White Oak Creek for a highway crossing. The lake was created in l943 when interlocking steel pilings were placed on the upstream side of the culvert. A vertical sliding gate, four by six feet in size, with the top elevation at 750 feet above mean sea level (msl) was used to control water level in the lake. At the overflow elevation of 750 ft msl, the volume of the impoundment was approximately 10.5 x 10^6 ft^3 (230,000 m^3) with a surface area of 44.2 acres (180,000 m^2). Another four-foot-square gate was fitted into the piling near its base; this was used to drain the lake in 1955. The lake was usually maintained at 748 ft msl, resulting in a lake surface area of approximately 35.9 acres (145,000 m^2). The historical changes that influenced the surface area of White Oak Lake are listed in Table 2.1.

Two small earthen fills were placed across White Oak Creek at mile 2.0 and 2.3 in 1944 to create preliminary settling basins for liquid waste from ORNL (Figure 2.6). These settling basins permitted the deposition of suspended radionuclides before they were returned to White Oak Creek. The lower fill caused the formation of an extensive marshy area of 35.87 acres (145,000 m^2). On September 29, 1944, the two earthen dikes were washed out during a storm that released 8.8 inches (22 cm) of rainfall in 26 hours (Morgan and Western, 1947; Setter and Kochtitzky, 1950). The Settling Basin, below the Gunite tank farm near White Oak Creek, overflowed for about 2 hours during this storm, but emergency stabilization activities prevented its sides from being washed away, and the enormous dilution caused by

the rain prevented the occurrence of significant radiation hazards downstream. There was concern that White Oak Dam might be breached, but just when it appeared that the dam would be overtopped, the water began to recede. To remove the threat of future washout of the Settling Basin, which might dump a thousand or more curies of activity into the drainage system, the stream bed of White Oak Creek near the Settling Basin was widened and deepened.

2.5.2 Estimates of Radioactivity in Bottom Sediments of White Oak Lake

The first discharges of radioactivity to White Oak Creek started on March 6, 1944, and the first measurements of radioactivity in the White Oak Creek drainage system were made by Overstreet and Jacobson (1944). On April 26, 1944, along the creek's course, activity ranged from the 10^{-6} Ci g^{-1} (37,000 Bq g^{-1}) in sediment samples just below the outfall in White Oak Creek to10^{-9} Ci g^{-1} (37 Bq g^{-1}) in Clinch River sediment just below the mouth of White Oak Creek. Because of the higher than expected levels of activity in the samples, the release of radioactivity to White Oak Creek was discontinued until the Settling Basin was completed**.**

2.5.3 Draining of White Oak Lake in 1955

By 1954, it was believed that White Oak Lake was in equilibrium with White Oak Creek in terms of its ability to dilute and otherwise retain radioactive material (Burnett, 1947). As a result, the fish populations were poisoned and removed, and the lake was drained in October 1955 (Lee and Auerbach, 1959). According to Browder (1959) White Oak Lake was drained for the following reasons:

- to perform maintenance work on the dam facilities;
- to destroy and dispose of the aquatic species in the lake;
- to avoid attracting and harboring migratory waterfowl;
- to provide additional safeguards by increasing retention potential;
- to facilitate and improve control of activity release; and
- to permit modification of sections of the lake area for research use.

Draining the lake was accomplished in a slow and deliberate manner to prevent disturbing the alluvial material and the silty areas. After the draining, White Oak Creek flowed in a narrow channel through the lake bed and through the gate in the dam, which could be closed in case of emergency (Photograph 3). A continuous water sampler and a radiation monitor had been installed at the dam in 1950**.** Although the monitor could detect a "slug" of radioactivity, which in turn would sound an alarm, it was considered a stop-gap instrument until a better one could be developed (Browder, 1959).

Table 2.1 Historical changes in White Oak Lake.

Date	Surface area (ha)	Events	References
1941		Highway fill and culvert installed by TVA	Smith 1945 as in Krumholz 1954
1943	14.5	Sheet piling dam installed with spillway with vertical sliding gate	Krumholz 1954
1943		Generation of radioactive waste at ORNL began and lake served as final settling basin (750 ft MSL)	Morton 1961
1944	NA	Dikes at White Oak Creek km 3.3 and 3.9 washed out (7.75 in., 26 in., 3.5 in. runoff)	Setter and Kochtitsky 1950
1945	12.2	Investigation of structural strength of dam (746.5 ft)	Oakes et. al. 1982a
1948	10.3	Lake lowered to 745.5 ft. to facilitate sediment sampling, normal operation from 1948 to 1955 was form 747 to 749 ft.	Oakes et. al. 1982a
1953	NA	Lake partially drained during rotenone survey of fish population	Oakes et. al. 1982a
1955	2.8	Lake drained: radionuclides in lake sediment and water believed to be in equilibrium so lake served no useful function in retaining radioactivity but could function as an emergency storage basin	Morton 1961
1956	0.4	Significant releases of ^{137}Cs probably from erosion of freshly exposed sediment after lake was drained	Lackey 1957
1959	NA	Gate structure renovated to prevent inflow of backwaters from Clinch River	Morton 1961
1960	3.2	Dam closed, surface level raised	Kolehmainen and Nelson 1969
1963	6.0	Completion of Melton Hill Dam	Kolehmainen and Nelson 1969
1967	8.1	None reported	McMaster 1967
1969	10.5	None reported	Kolehmainen and Nelson 1969
1979	4.6	Lake level gradually dropped from 745 to 742 ft. because of potential instability of the dam	Oakes et. al. 1982a
1980	6.9	Construction of a berm to stabilize dam was completed	Boyle et. al. 1982
1983	6.9	Discharge channel and weir constructed, roadbed rerouted	Oakes et. al. 1982b
1988	6.9	Estimate of surface area and volume (43,900 m^3) at lake elevation of 745 ft.	Cox et. al. 1991

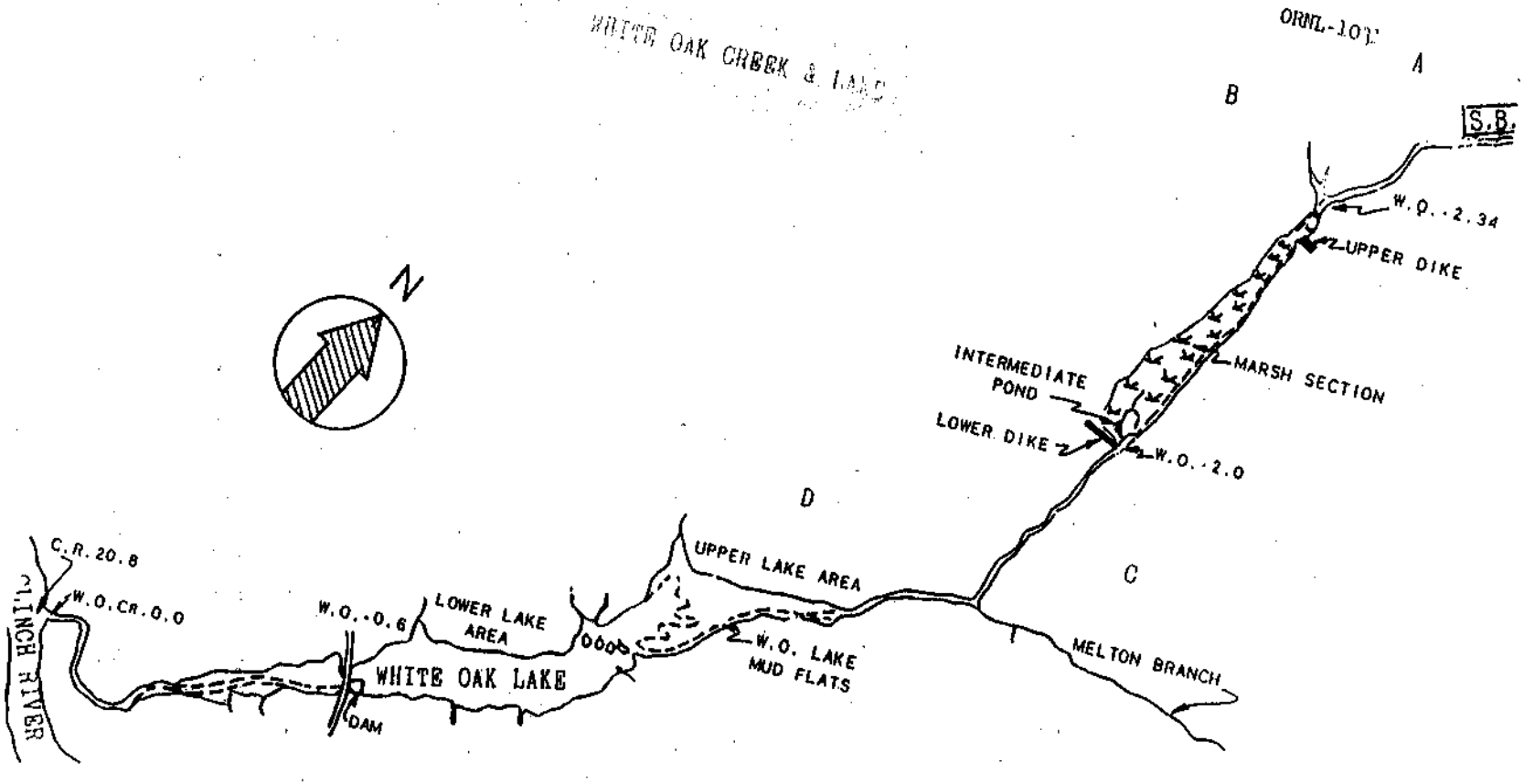


Figure 2.6 An early drawing showing White Oak Lake, the upper and lower earthen fills (dikes), and the Intermediate Pond. The upper and lower dikes were "washed out" when 8.8 inches of rain reportedly fell on the White Oak Creek Watershed during a 26-hour period starting on September 29, 1944.

2.5.4 Flooding of White Oak Lake Bed in 1956

Heavy rains in early 1956 flooded the former lake bed before it was revegetated. The greatest annual release of ^{137}Cs (170 Ci or 6.3×10^{12} Bq) from White Oak Lake was reported for 1956 (Ohnesorge, 1986). Measurements of the depth of the sediment in the lake in 1961 were compared with those made by the Tennessee Valley Authority in 1953. The measurements indicated that the channel of White Oak Creek, which had been filled with sediment before the lake was drained, was eroded back to its preimpoundment depth. Sediment depth in the lower one-third of the lake was approximately the same, but the middle and upper two-thirds of the lake had slightly less sediment than reported in 1953. An estimated 4250 m^3 (150,000 ft^3) of silt had been eroded from the lake bed between 1953 and 1961 after draining of the lake (Lomenick et al., 1961; Oakes et al., 1982a). In 1960 the spillway gates were reworked and closed to prevent the backflooding of water through the dam (Kolemainen and Nelson, 1969).

2.5.5 Ruthenium Releases 1959-1961

Ruthenium seeped from the waste pits onto the former bed of White Oak Lake (the area exposed following the draining), where the ruthenium was either sorbed onto the soil or transported by water over White Oak Dam. It was believed that much of the ^{106}Ru would be retained in the soil or sediments of the upper lake bed during the dry season, and because of its short physical half-life (372 days), ^{106}Ru releases past White Oak Dam would not be problematic. Lomenick (1963) estimated that less than half of the ^{106}Ru leaving the pit system during 1959 and 1960 was detected at White Oak Dam. Through December 1961, the pit system had received a total of about 22,000,000 gallons (83,000 m^3) of waste containing approximately 235,000 Ci (8.7×10^{15} Bq) of ^{106}Ru; 183,000 Ci (6.8×10^{15} Bq) of ^{137}Cs; 43,000 Ci (1.6×10^{15} Bq) of ^{90}Sr; 77,000 Ci (2.8×10^{15} Bq) of rare earths exclusive of ^{90}Y and lesser amounts of ^{103}Ru, ^{89}Sr, ^{60}Co, and ^{125}Sb.

According to Lomenick (1963) a reasonable estimate of the quantity of ^{106}Ru in the lake bed as of February 1962 was 1,200 Ci (4.4×10^{13} Bq), but larger quantities could have been in the lake bed at different times. Estimates of the amount of ruthenium leaving the pits represented values based on continuous records of stream flow and weekly grab samples in the stream flow area and, at best, represented only reasonable estimates. During the dry summer months, the streams that drained the waste pit area had low flow rates, and the seepage from the pits mainly recharged the groundwater. However, in the wet winter season, when stream flow was highest and the groundwater beneath the lake bed was near the surface, more of the waste water flowed over the lake bed into the creek. It was during the wet season that the greatest quantity of ruthenium was expected to be released at White Oak Dam. It was also during high water when the water level was above the pilings at the dam that monitoring of the flow through White Oak Dam was the most inaccurate.

2.5.6 Estimates of Radionuclide Concentrations in White Oak Lake Sediment

Estimates of the radioactivity in the bottom sediment of White Oak Lake were made by Abee (1953) for 1950, 1951, and 1952 (Table 2.2). These estimates were based on core samples that were collected from the lake and analyzed for radioactivity. In most cases, only the top 3 inches were analyzed; however, some cores were analyzed to greater depths. Abee compared his estimates of the total radioactivity in the sediment to estimates by others for 1945, 1946, and 1948 (Table 2.2). In 1945 and 1946, the total activity was listed as approximately 20 Ci (7.4×10^{11} Bq), despite higher sediment activity values reported in an earlier survey (Cheka and Morgan, 1947). The estimated inventory in the sediment for 1948 was 310 Ci (1.1×10^{13} Bq), which was in line with the estimates for 1950 through 1952.

Table 2.2 Total Activity in White Oak Lake Sediment 1945-1952 (Abee, 1953)

Annual Assay	Total Activity in Sediment
1945*	21 Ci (7.8×10^{11} Bq)
1946*	20 Ci (7.4×10^{11} Bq)
1948*	310 Ci (1.1×10^{13} Bq)
1950	392 Ci (1.5×10^{13} Bq)
1951	359 Ci (1.3×10^{13} Bq)
1952	303 Ci (1.1×10^{13} Bq)

*estimates from other sources

The most comprehensive analysis of radionuclide concentrations in White Oak Lake sediment was conducted by Lomenick and Gardiner (1965). The inventory of radionuclides in the lake sediments was based on 250 sediment core samples taken in 1962. Ninety-three cores were analyzed for ^{90}Sr and trivalent rare earths (TREs), and 250 cores were analyzed for ^{137}Cs, ^{106}Ru, and ^{60}Co. A total of 1,038 Ci (3.8×10^{13} Bq) of ^{106}Ru, 704 Ci (2.6×10^{13} Bq) of ^{137}Cs, 152 Ci (5.6×10^{12} Bq) of ^{60}Co, 16.1 Ci (5.9×10^{11} Bq) of TREs, and 14.6 Ci (5.4×10^{11} Bq) of ^{90}Sr were estimated to be in the sediment of White Oak Lake (Lomenick and Gardiner, 1965). Approximately 60% of the activity of each radionuclide was found in the upper 0 to 6 inches of sediment, with decreasing amounts found from 18 to 24 inches (Table 2.3). Ninety percent of the activity was composed of ^{106}Ru and ^{137}Cs, both of which decreased with depth. The lake bed contained about 1,000,000 ft^3 (28,000 m^3) of contaminated sediment, which was as much as 2 ft (0.6 m) thick in the lower part of the lake near the dam.

2.6 Clinch River 1943-present: The Clinch River Study

Although the Clinch River had been analyzed on a limited basis before 1959, the Clinch River Study from 1959 to 1964, was much more comprehensive and produced a large amount of usable data. The Clinch River Study was organized in 1959 as a cooperative project. Four Federal agencies (Tennessee Valley Authority, US Atomic Energy Commission, US Geological Survey, and US Public Health Service), two State of Tennessee departments (Tennessee Department of Public Health and Tennessee Game and Fish Commission), and the Oak Ridge National Laboratory were joint participants in the planning and administration of the study. A program of study was drawn up in 1959, and implemented over five years (Struxness et al., 1967).

The initial objectives of the Cinch River Study included: (1) to determine the fate of radioactive material being discharged into the Clinch River; (2) to determine and understand the mechanisms of dispersion of radionuclides in the river; (3) to evaluate the direct and indirect hazards of waste disposal practices; and (4) to evaluate the usefulness of the Clinch River for radioactive waste disposal. During the study, Melton Hill Dam was being constructed on the Clinch River a short distance above the mouth of White Oak Creek to serve as a peaking unit for the production of electrical power. The Clinch River Study was also designed to determine the influence of Melton Hill Dam operations on the distribution and concentration of radionuclides in the Clinch.

Six Clinch River Study status reports, edited by R. J. Morton, and a number of supplementary reports resulted from the study. One major part of the study involved the sampling and analysis of sediment core samples from the Clinch and Tennessee rivers. The ensuing report emphasized the transport and distribution of radionuclides in the sediment of the Clinch River. The locations of sampling stations that were used in the Clinch River Study were as follows (Churchill et al., 1965):

Table 2.3 Quantity and distribution of radionuclides in White Oak Lake bed – December 1962 (Lomenick, 1963).

Radionuclides	Depth from surface (in.)									
	0 to 6		6 to 12		12 to 18		18 to 24		Totals	
	Curies	%	Curies	%	Curies	%	Curies	%	Curies	%
^{106}Ru	594 ± 76*	57	276 ± 38	27	112 ± 19	11	56 ± 10	5	1038 ± 88	100
^{137}Cs	468 ± 20	66	204 ± 25	29	29 ± 12	5	3 ± 1	<1	704 ± 35	100
^{60}Co	119 ± 9	78	22 ± 2	15	8 ± 1	5	3 ± 0.5	2	152 ± 9	100
TRE**	13 ± 3	78	2.5 ± 1.0	15	1.0 ± 0.7	6	0.1 ± 0.1	1	16.6 ± 3.3	100
^{90}Sr	10 ± 0.9	68	3.5 ± 0.5	24	1.0 ± 0.3	7	0.1 ± 0.1	1	14.6 ± 1.1	100

* Numbers following ± are estimated standard errors.
** Trivalent rare earths exclusive of ^{90}Y.

Station 1. Clinch River at Oak Ridge water plant -- Clinch River Mile 41.5
Station 2. White Oak Creek at White Oak Dam -- White Oak Creek Mile 0.6
Station 3. Clinch River at Gallaher Bridge -- Clinch River Mile 14.5
Station 4. Clinch River above Centers Ferry -- Clinch River Mile 5.5
Station 5. Tennessee River at Loudon, Tennessee -- Tennessee River Mile 591.8
Station 6. Tennessee River at Watts Bar Dam -- Tennessee River Mile 529.9
Station 7. Tennessee River at Chickamauga Dam -- Tennessee River Mile 471.0

Sampling began in November 1960 and extended through November 1962, except at the Gallaher Bridge where sampling began in January 1962 and was discontinued at the end of November 1962. Samples were analyzed by the US Public Health Service, Cincinnati, Ohio, as well as by the Oak Ridge National Laboratory. The Clinch River Study provided data on the concentrations of radionuclides in water, sediments, and biota.

2.7 Waste Discharged Directly into Creeks 1950-present

Several comparatively minor sources of radioactive waste were not included in the main ORNL liquid waste control systems. These sources produced relatively small quantities of low-activity waste, most of which was released to the nearest creek within the White Oak Creek drainage. These sources are described briefly below.

The Low Intensity Test Reactor (LITR). The LITR was located within the ORNL-controlled access area, less than 100 ft (30 m) north of the ORNL Graphite Reactor. The LITR began operating in February 1950. In 1953, the power level was increased, and it was utilized as a general research facility until it was decommissioned in October 1968. The LITR discharged "highly" radioactive waste to a monitoring tank in the "highly" radioactive chemical waste system. Two retention ponds received the coolant water when the reactor was drained or when it leaked coolant. Cooling water from irradiation experiments in the reactor also was discharged to these ponds. Most of the radioactivity was in the form of ^{24}Na, which has a half-life of about 15 hours. The ponds were periodically drained to a branch of White Oak Creek after the activity in the water had decayed to less than 100 gross beta and gamma counts/minute/milliliter. The ponds are no longer in use (Coobs and Gissel, 1986).

The Oak Ridge Research Reactor. The Oak Ridge Research Reactor ("swimming pool reactor") was located in Bldg. 3042 and operated at 30 MW from May 1958 until July 1987, when it was shut down. The LITR retention ponds also received reactor water drainage from the "warm" sump in the Oak Ridge Research Reactor building. "Highly" radioactive waste was handled by the same tanks that served the LITR. The retention ponds were periodically drained into a branch of White Oak Creek.

The Homogeneous Reactor Test (HRT). The HRT was located in Melton Valley in the 7500 area across a high ridge from the main X-10 area. A retention pond (Pond 7556), which overflowed to Melton Branch, received reactor storage-canal water. The capacity of the pond equaled the entire volume of all the HRT building cells. If activity was too high, the water was drained into a 12,000-gallon (45-m^3) stainless steel tank buried near the pond. Waste was then hauled to the "highly" radioactive waste disposal pits (Browder 1959). Operation of the first HRT began in February 1953 with a small (150 kW) reactor that was dismantled in 1954. A second HRT was started in January 1958. It was designed to operate at 5 MW but was plagued with problems and shut down early in 1961. The waste pond is no longer in service and has been filled and covered with asphalt. No records of its contents are available. However, approximately 10% of the ^{90}Sr entering Melton Branch reportedly originates from the contaminated flood plain adjacent to the covered pond (Duguid et al., 1977). The flood plain soils also contain higher than background concentrations of ^{239}Pu (Duguid et al., 1977).

The Aircraft Reactor Test (ART). The ART was located near the HRT site in Melton Valley. The first test run occurred in October 1954, and the ART was canceled in September 1957. The facilities of the ART drained into a sump from which low-level waste was pumped into a retention pond. This pond was excavated in the shale near Melton Branch about 300 yards (270 m) from the reactor building. No provisions were made for the overflow of the pond, which has not been used since the 1950s (Coobs and Gissel, 1986).

The Tower Shielding Facility (TSF). The original Tower Shielding Reactor was completed in 1954 near the Clinch River about 2 miles (3.2 km) from the main Laboratory. The original reactor was replaced by a second reactor in 1960 at the same site. An open pool 20 ft (6 m) square and 25 ft (7.6 m) deep and a storage basin 24 ft by 8 ft by 12 ft deep (7.3 m $\times$ 2.4 m $\times$ 3.6 m) served these reactors. Water circulates through one of the reactors and returns to the pool, which periodically overflows downhill away from the facility. A one-gallon sample is taken from the pool once a week for radiation analysis, and mud on the hillside is sampled when an overflow occurs. Sodium-24 and irradiated impurities comprise almost all the radioactive contamination in the water. Four 20,000-gallon (76-m^3) tanks are available at the TSF site for contaminated water holdup if a fuel element ruptures (Coobs and Gissel, 1986).

Central Facility Reactors. In 1986, three reactors were operating at the central facility at ORNL: The Oak Ridge Research Reactor, started in 1958 and shut down in July 1987; the Bulk Shielding Reactor, completed in 1950; and the Pool Critical Assembly (PCA), which was a 10 kW nuclear assembly located in the corner of the Oak Ridge Research Reactor pool. The three reactors had a common water purification system and a common heat dissipation system.

The High Flux Isotope Reactor (HFIR). The High Flux Isotope Reactor, which was completed in 1966, is located in Building 7900 in Melton Valley. The HFIR is a 100 MW pressurized light water reactor that uses enriched uranium (93%) fuel. The HFIR's main purpose is the production of research quantities of transplutonium elements. Gaseous releases are discharged through the Bldg. 7911 stack, and liquid effluents containing radionuclides are piped to the radioactive waste treatment facilities in Bethel Valley (Boyle et al., 1982). A retention pond at the HFIR facilities was part of the low-level waste system; this has been used in the past to store activation products and is apparently still used at times (Coobs and Gissel, 1986). The liquid was pumped from the pond after decay or settling of suspended radionuclides. The HFIR was temporarily shut down in November 1986 because of concern regarding vessel embrittlement and resumed operation at 85% of its original power in 1990.

2.8 Solid Waste Burial Grounds (SWSAs)

Six solid waste burial grounds (also called solid waste storage areas or SWSAs) have been used at ORNL since 1944 (Burch et al., 1972). These burial grounds were numbered consecutively in the order in which they were first used (Figure 2.7, Table 2.4). Burial Grounds 1 and 2 were located for convenience near the ORNL complex in Bethel Valley. Burial Ground 3 was located to the west, out of sight of the main ORNL complex in Bethel Valley. Burial Grounds 4, 5, and 6 were located in Melton Valley, which is separated from Bethel Valley by Haw Ridge. During the operation of Burial Grounds 1, 2, and 3, waste was placed into trenches and backfilled. Few historical records are available for these burial areas, and records that were kept for burial grounds 3, 4, and 5 were accidentally destroyed by fire in 1961 (Webster, 1976).

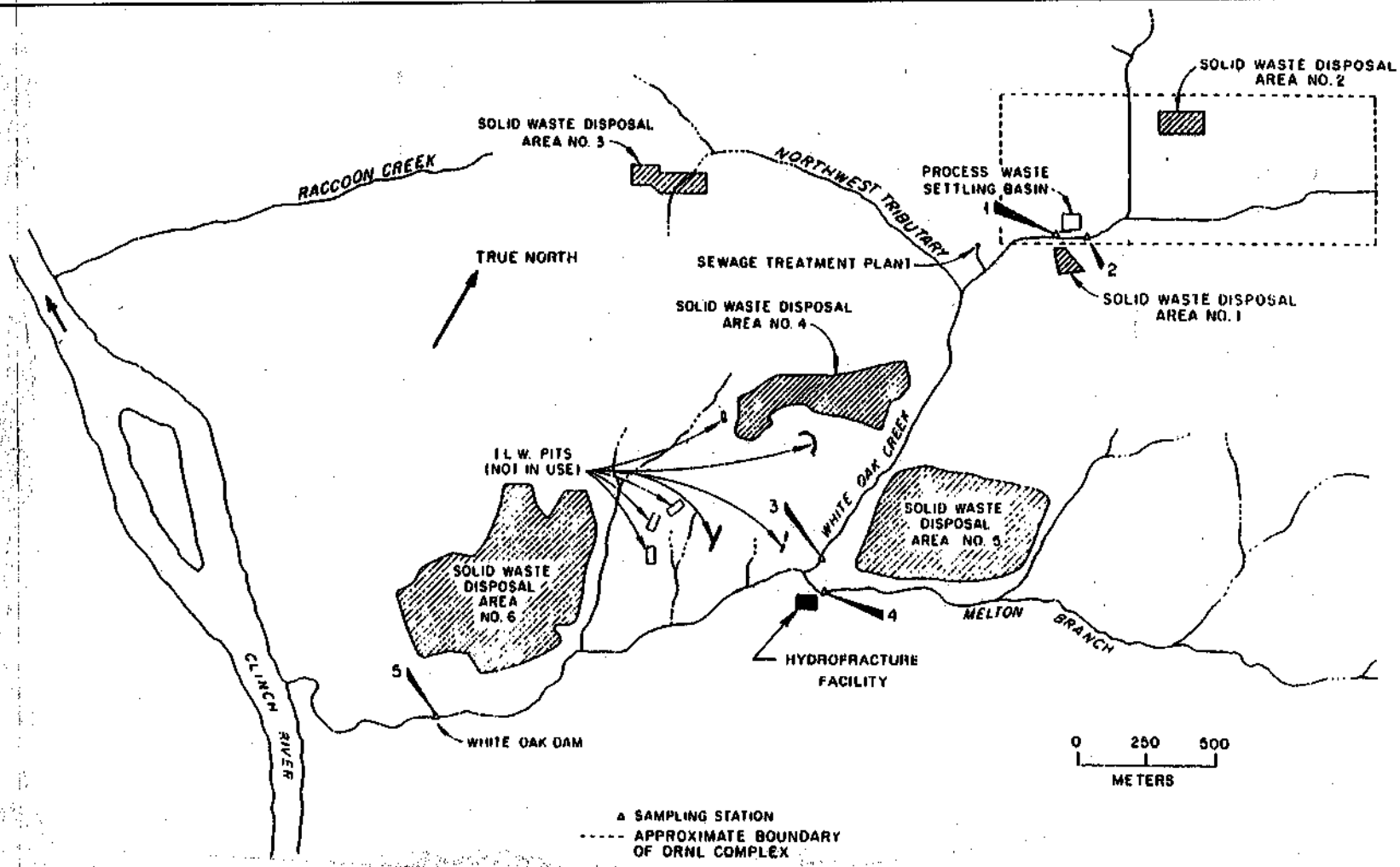


Figure 2.7 A map showing the location of the 6 solid waste disposal areas (SWSAs) in relation to the main Oak Ridge National Laboratory complex, the seepage pits, and the potentially influenced streams.

From 1955 to 1963, ORNL served as the Southern Regional Burial Ground of the Atomic Energy Commission. Solid waste from other sites such Argonne National Laboratory, Knolls Atomic Power Laboratory, Mound Laboratories, Battelle Memorial Institute, General Electric Company in Evansdale, Ohio, and other off-site installations contributed a large fraction of both the material and the radioactivity buried in ORNL's solid waste disposal areas. Radioactive solid wastes from these sources occupied approximately one-half of the burial ground space in 1959 (Browder, 1959). Compared to the liquid wastes previously described, the solid wastes usually contained minor quantities of radioactivity, probably about 1% of the total disposed of at ORNL. Accurate determination of the curies handled was not possible, because the wastes were nonuniform mixtures of many kinds of materials. Records for most solid waste burial grounds are scant (Webster, 1976).

Table 2.4 Solid Waste Burial Grounds, Periods of Operation and Sizes

Location/Name	Dates of Operation	Size: Acres	Size: Hectares
Bethel Valley			
Burial Ground 1	1944-1944	1	0.4
Burial Ground 2	1944-1946	3.6	1.5
Burial Ground 3	1946-1951	7	2.8
Melton Valley			
Burial Ground 4	1951-1959	23	9.3
Burial Ground 5	1958-1973	33	13.3
Burial Ground 6	1973-present	70	28.3

Burial Ground 1 is located at the foot of Haw Ridge at the edge of the Laboratory complex and is about 25 ft (8 m) south of White Oak Creek. Trenches were used to dispose of contaminated broken glassware and other contaminated equipment. Reportedly, the burial ground was abandoned when water was found in one of the trenches. Analysis of water from one of the monitoring wells revealed a small quantity of ^{90}Sr (Duguid, 1975).

Burial Ground 2 covers about 2 to 3 (8,000-12,000 m^2) acres close to the Graphite Reactor and the chemical separation plant on the lower half of a hill located near the east entrance to ORNL. Webster (1976) reported that solid waste was placed in black iron drums and buried in trenches. Liquid waste contaminated by plutonium was put in stainless steel drums and either buried or stored in a ravine. In addition, waste from an off-site source was buried and covered with concrete near the location of a transformer station. In 1945, two shipments of off-site waste, heavily contaminated with polonium, were buried under a concrete slab. SWSA 2 (Solid Waste Storage Area 2) was closed in 1946 because it was considered to be incompatible with long-range land-use planning at the Laboratory. Following closure of SWSA 2, the stainless steel drums containing plutonium-contaminated liquid waste were transferred to SWSA 3. In SWSA 2, beta and gamma waste buried in iron drums, which had deteriorated, was carefully removed and reburied in SWSA 3. The hillside was backfilled and contoured between 1946 and 1949, and SWSA 2 is currently neither fenced nor marked so that its location cannot be readily identified (Coobs and Gissel, 1986).

Burial Ground 3 is about 0.6 miles (1 km) west of the west entrance to the Laboratory complex. It is a flat, forested area at the foot of Haw Ridge on the north side. The site was supposedly chosen because it was close to the Laboratory yet out of sight. The burial ground became operational in April 1946, although it had received "hot material" from off-site waste producers prior to this date. The alpha waste from SWSA 2 was removed and encased in concrete in SWSA 3. In SWSA 3 trenches were cut parallel to each other across the width of the area, and alpha-contaminated waste and beta-gamma waste were dumped into the trenches. Beta- and gamma-emitting wastes were buried in separate, unlined trenches and backfilled with the excavated soil (Coobs and Gissel, 1986).

A groundwater divide underlies the western part of the Burial Ground 3 site. Easterly ground water movement enters a tributary of White Oak Creek. Western ground water movement is toward Raccoon Creek. Surface runoff is directed to White Oak Creek via shallow drainage ditches. SWSA 3 was closed in 1951 after about seven acres (28,000 m^2) had been utilized. It is currently fenced and shows no sign of erosion.

Burial Grounds 4, 5, and 6 were situated in Melton Valley based on a recommendation by Professor P. B. Stockdale, Head of the Department of Geology and Geography at the University of Tennessee, Knoxville. After studying the geology and hydrology of the X-10 site, Stockdale (1951) recommended that all future contaminated solid waste be buried in the Conasauga shale belt of Melton Valley to prevent the contamination of the limestone in Bethel Valley, which seemed inevitable if waste were buried there.

Burial Ground 4 was opened in 1951 at the foot of Haw Ridge along Lagoon Road and near the flood plain of White Oak Creek. The site recommended by Stockdale was northwest of the lower end of White Oak Lake; but, apparently, because of the closeness to the Laboratory and because it was underlain by Conasauga shale, the site at the foot of Haw Ridge was selected. "Higher-level" waste and some "special high-level" waste were placed in auger holes 1 to 2 feet (0.3-0.6 m) in diameter and approximately 15 feet (4.6 m) deep (Lomenick and Cowser, 1961). Although records are not readily available, in 1957 and 1958 approximately 7,300 m^3 (255,000 ft^3) and 9,500 m^3 (336,000 ft^3), respectively, were reportedly buried in SWSA 4. During this time ORNL produced approximately 50% of the waste, while other Oak Ridge installations and off-site generators contributed the remainder. Argonne National Laboratory, Knolls Atomic Power Laboratory, Mound Laboratory, and the General Electric Company of Evansdale, Ohio, were the principal off-site shippers, but over 50 agencies also used SWSA 4 (Lomenick and Cowser, 1961). Subsequent to closure of the 23-acre (93,000-m^2) site in 1959, much of the site was covered by uncontaminated fill, which raised the land surface as much as 20 feet (6 m) in that area and also raised the groundwater level.

Both alpha and beta activity were found in water discharged from seeps originating from SWSA4. White Oak Creek samples taken above and below the burial ground showed that the radionuclide load originating from this burial ground was comparatively small compared to the radiochemical effluents discharged from the Laboratory.

Burial Ground 5 consists of two sections on the hillside east of the confluence of Melton Branch and White Oak Creek. The South Area of SWSA 5 contains most of the buried waste and more closely fitted the criteria that had been developed for a solid waste disposal site. The North Area of SWSA 5 was appended later to provide retrievable storage of certain fissile alpha waste. Detailed geologic and hydrologic investigations were conducted on the site. SWSA 5 was used for the disposal of all types of wastes starting in 1958; however, by far the largest volume of material was general radioactive waste, which was buried in trenches in “semi-retrievable" form (Duguid et al., 1977). Alpha-containing wastes were inserted in the lower part of the area of "undefined trenches" and capped with concrete, whereas beta-gamma contaminated wastes were simply covered with weathered shale previously removed by the

excavation. Auger holes were used for disposal of higher level wastes and certain less hazardous materials to better utilize all suitable land.

Both groundwater and surface water drainage from SWSA 5 are predominantly southeast toward Melton Branch with some southwest movement toward White Oak Creek. As a result, most of the radionuclide transport by surface water is monitored at the Melton Branch Station. Water collected from seeps from SWSA 5 in 1974 indicated that ^{90}Sr and ^{3}H were principal contaminants. Several thousand curies of ^{3}H, believed to have originated in groundwater from SWSA 5, pass the Melton Branch monitoring station annually (Webster, 1976).

All the retrievable waste is located in SWSA 5, including above-ground storage. The appreciable decline in the volume by 1963 was caused by the discontinuation of this site as the Southern Regional Storage Area. SWSA 5 was closed in 1973.

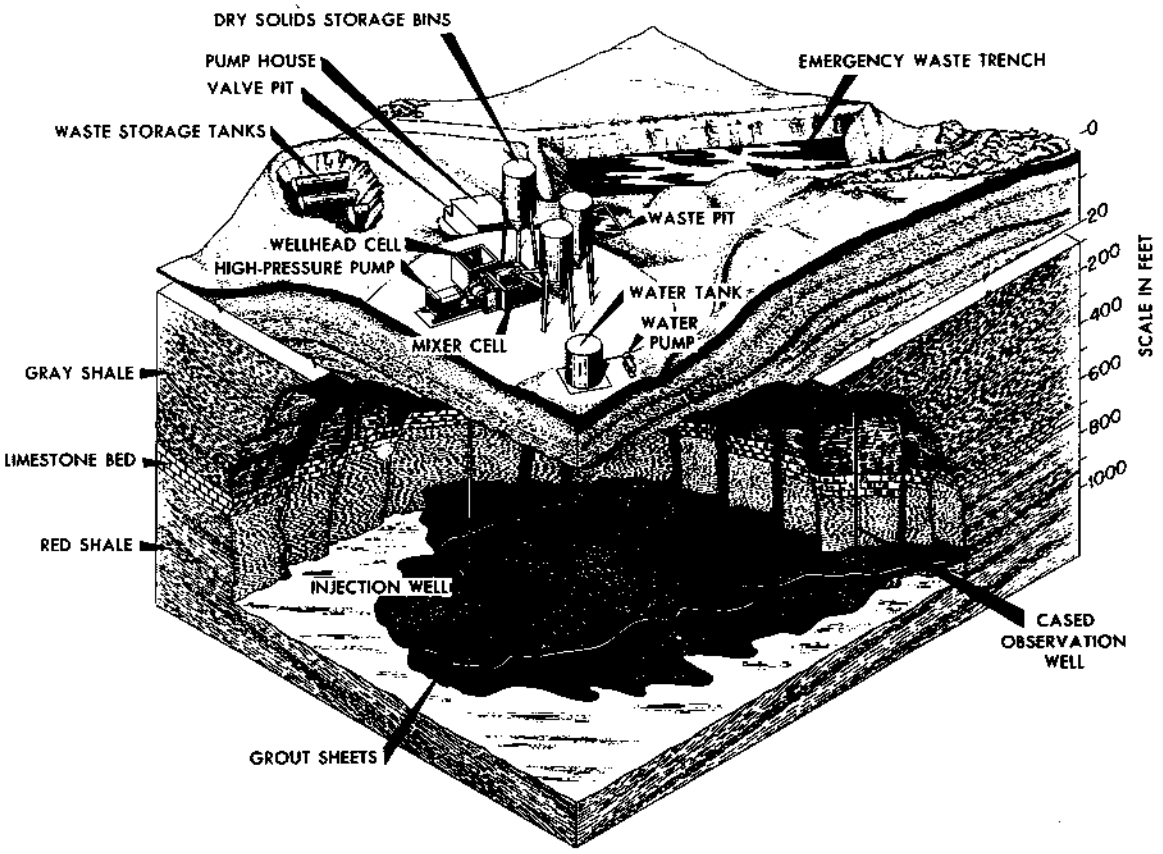
Burial Ground 6 is located immediately northwest of White Oak Lake on a wooded hillside and is approximately 70 (280,000 m^2) acres in size. The hydrology of the site was generally similar to that of Burial Ground 5. The first waste was buried at the site in 1969, although it did not become the principal burial site for Laboratory waste until 1973, when SWSA 5 was closed. Surface runoff from the SWSA 6 site drains directly to White Oak Lake or to short drainages that discharge into White Oak Lake. Trenches on the hillside, generally in series of 30 to 42, were excavated where the water table is 10 to 15 feet (3 to 4.5 m) below the land surface during its seasonal high.

2.9 Tritium Releases from Seepage of the Burial Grounds

The principal radionuclides involved in groundwater seepage from the ORNL burial grounds since the early 1960s are ^{3}H and ^{90}Sr (Duguid et al., 1977). Tritium has been observed at White Oak Dam for many years. Starting in 1967 there was a dramatic increase in the quantity of tritium released into White Oak Creek due to shipments of material received from Mound Laboratory prior to 1967. The waste had been disposed of in SWSA 5, and the bulk of the tritium was discharged from there. Approximately 90% of the tritium released to White Oak Creek was monitored at the Melton Branch station above the confluence with White Oak Creek. Most of the ^{90}Sr also was discharged from SWSA 5 and monitored at the same station on Melton Branch.

2.10 Melton Valley Hydrofracture Facilities

In December 1966, hydrofracturing was selected instead of waste pits as the method for disposing of intermediate-level liquid waste. In the hydrofracturing process, hydraulic pressure was used to initiate the formation of a crack between the layers of shale (Figure 2.8). An alkaline intermediate-level waste solution was mixed with a solid blend composed of cement and other additives; this mixture was injected under pressure into the crack in the impermeable shale formation at a depth between 700 and 1,000 ft (200 to 300 m). As the injection continued, the groundwater contamination originating from the hydrofracture operations (Ohnesorge, 1986; USDOE, 1988).



ORNL FRACTURING DISPOSAL PILOT PLANT

Figure 2.8 A drawing of the Oak Ridge National Laboratory's hydrofracture plant located in Melton Valley.

2.11 Monitoring History for Releases of Liquid Radioactive Wastes

2.11.1 Settling Basin

To monitor the radioactivity released from the Settling Basin into White Oak Creek, the total volumetric discharge to White Oak Creek had to be measured and a representative sample collected and analyzed for radioactivity. Prior to 1948 the Settling Basin was grab-sampled periodically. From 1948 to 1950 the Settling Basin was grab-sampled every four hours during the work week. Flow measurements were made at the inflow to the Settling Basin by means of a v-notch weir and a head-height measuring device. Beginning in November 1950, flow measurements were made by a single 90^{o} v-notch weir in conjunction with a standard stilling well and Stevens water level or head recorder. A trebler proportional sampler was installed in the weir box; this sampler was equipped with a revolving dipper to collect a volume proportional to the flow through the weir at the time of each dipping and with a timer to limit the dips per hour.

2.11.2 White Oak Dam

Starting in 1944, water leaving White Oak Dam was sampled once a day during the work week by taking an 8-gal (0.03-m^3) grab sample. More frequent samples were taken when significant changes in activity were anticipated (Feige et al., 1960). About 140 mL of each daily sample, collected from the Settling Basin and White Oak Dam, were composited separately into monthly samples. The composited samples were evaporated and the gross beta activity determined for each location.

The methods and frequency of sampling the water discharged from White Oak Dam have changed over the years. From September 1944 to February 1945, monthly determinations were made of the average distribution of fission products in the water leaving White Oak Lake (Morgan and Western, 1947). The analyses were made by the Analytical Section of ORNL on a composite sample accumulated during the month. The method included separation of the principal long-lived fission products, evaporation to dryness, and measurement of the beta activities of the separate elements. Routine monthly analyses of individual elements were discontinued in January 1945, because the concentrations of individual elements were too low for accurate measurements and the concentrations of strontium and barium were considered below levels that could constitute individual hazards from ingestion. Subsequently, monthly samples were counted for gross beta activity only. However, an analysis of individual fission products was made if there was an anomolously high gross beta count.

In 1949 improved radiochemical analysis techniques enabled reporting of individual radionuclide concentrations (Struxness et al., 1967). Radiochemical separation was performed on the monthly composite samples; the samples were analyzed for the trivalent rare earths and isotopes of ruthenium, zirconium, niobium, cesium, strontium, iodine, barium, and cobalt. Cerium was extracted from a trivalent rare earth sample, and a beta absorption measurement was made for ^{89}Sr and ^{90}Sr. Analysis of ^{131}I began in 1948 or 1949, but presumably on a weekly basis because of its short physical half-life (8.05 days). Analyses of monthly composites for cobalt were started by 1955, and for tritium by 1964 (Webster, 1976).

After 1949 “highly” radioactive waste was not deliberately dispersed into the uncontrolled environment (i.e., White Oak Creek) (Feige et al., 1960). Prior to 1949, however, the supernatant from the Gunite storage tanks was mixed with the Laboratory's process waste waters and discharged through White Oak Creek. However, White Oak Lake inflow radioactivity measurements did not match outflow radioactivity measurements, and from 1954 to 1960 more curies left White Oak Dam than could be accounted for by

the inflow from the Settling Basin (Feige et al., 1960). According to Feige et al. (1960) the monthly sample did not represent the average monthly activity released from White Oak Dam because a constant sample volume was set aside each day for a composite radioactivity analysis, without regard to variation in volumetric discharge rates. The method discriminated against days of higher flow when large amounts of activity were discharged through White Oak Dam.

Prior to the draining of White Oak Lake in October 1955, a gauging station at the dam was used to measure flow rates. For about four years after the lake was drained, the flow from White Oak Dam was estimated by summing the discharges at stream flow stations on White Oak Creek below ORNL and on Melton Branch and by using an adjustment factor of 1.16 to compensate for the local inflow downstream (Struxness, 1962). Grab-sampling at the dam continued during this period to measure the radioactivity in the water.

During June 1958, a continuous, nonproportional sampler, a scintillation probe to measure gross beta counts, and a remote recorder were installed at White Oak Dam. Daily gross beta determinations of the Settling Basin effluent fluctuated from 30 to 3000 counts/minute/milliliter and at White Oak Dam from 2 to 200 counts/minute/milliliter. These fluctuations introduce a high degree of uncertainty in the results obtained by the grab-sampling method. The number of curies of individual radionuclides released per year was derived by multiplying the yearly discharged quantity of gross beta activity (determined by laboratory analysis) by the percentage of activity for each radionuclide found in the monthly composite samples. This calculation continued to be inaccurate because the daily samples were not collected in volumes proportional to the daily flow, i.e., a constant amount of each daily grab sample (140 mL) was included in the monthly composite sample (Struxness, 1962).

The nonproportional sampler was replaced in 1960 with a continuous proportional flow monitor that took samples in proportion to discharge (Webster, 1976). These composite samples were collected weekly and analyzed for gross beta activity to evaluate the gross concentration of radioactivity entering the Clinch River. The samples were also analyzed for transuranic alpha emitters, total strontium, and ^{131}I. Monthly composites, made from portions of the weekly samples, were analyzed for gross beta, ^{90}Sr, ^{131}I, ^{137}Cs, ^{106}Ru, ^{60}Co, tritium, and transuranic radionuclides. From Nov. 13, 1960, to Dec. 1, 1962, a continuous, daily, automatic collection of water in proportion to the flow was made at White Oak Dam (Struxness et al., 1967).

According to Duguid et al. (1977), the maximum measurable discharges at monitoring stations 3, 4, and 5 (White Oak Creek, Melton Branch, and White Oak Dam, respectively) at the time were 50 cfs, 19 cfs, and 150 cfs (1.4, 0.5, and 4.2 $m^3 s^{-1}$), respectively. When stream flows exceeded these values at the respective sites, water flowed over the tops of the stations' weirs, and no reliable record of flow rates or volumes was obtained; furthermore, proportionality of collected samples was lost. Although discharges exceeded measuring capacities at these stations for only a small percentage of the time each year, abundant evidence indicates that a disproportionately large volume of water and sediment were transported during times of high flow volumes. Because radionuclides are transported in the water and in association with sediment, records of contaminant transport and release for these periods of high discharge will have larger uncertainties than data from lower flow periods.

Errors could have been introduced into the measurements of radionuclides under high-flow conditions in three ways: (1) sample volumes were no longer proportional to discharge when flows were sufficiently large to go over the top of the weirs; (2) sediment in the sample could have been unrepresentative of the type and quantity of the sediment in the discharge; and (3) subtle changes in the characteristics of the weirs and backwater areas could have occurred that might have caused weir ratings to lose their validity.

2.12 Summary

Section 2 of this report has described the historical management of liquid radioactive wastes at the X-10 site, historical releases of radioactive liquid wastes past White Oak Dam to the Clinch River, and the associated monitoring practices. This information was used in the estimation of yearly releases of radionuclides to the Clinch River (Section 5). Three periods in particular were identified as contributing large uncertainties to release estimates: (1) 1944-1948, when only gross beta counts were reported; (2) 1956-1959, when White Oak Lake was drained; and (3) 1959-1963, when large quantities of ^{106}Ru seeped from the waste disposal pits.

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3.0 PRELIMINARY SCREENING ANALYSIS FOR RADIONUCLIDES IN CLINCH RIVER WATER AND SEDIMENTS

Task 4 of the Oak Ridge Dose Reconstruction is charged with evaluating the impacts of all radionuclides released off-site from the Oak Ridge National Laboratory via White Oak Lake and White Oak Creek to the Clinch River. In February 1996, a conservative screening analysis was conducted for these radionuclides to identify any radionuclides or exposure pathways for which the dose or risk to off-site individuals was clearly below a level of concern warranting further detailed study. The purpose of this screening analysis was to focus the detailed risk assessment, described in the rest of this report, on the most important radionuclides and exposure pathways. Detailed study for contaminants whose presence is clearly below a minimum level of concern (Section 3.1.6) is not warranted, as further investigation is expected to show that the risk to any actual individual would have been much less than that calculated during the conservative screening analysis (Thiessen et al., 1996). The designation of a low priority to these radionuclides or exposure pathways permitted Task 4 resources to be focused on those situations that were most important in terms of dose or risk to exposed off-site individuals or populations. Radionuclides and pathways with a screening index above the guide have been investigated in more detail in later stages of Task 4; these analyses, including preliminary uncertainty analyses of doses and risks, are described in Sections 4-15 of this report.

3.1 Methodology for Screening

The intent of the conservative screening calculations described in this section was to produce screening-level estimates of risk to specified target individuals that are not likely to be underestimates of the actual risk to any exposed individual. In other words, the calculated screening index (a conservatively biased estimate of lifetime risk to the most at-risk individual) for each radionuclide or exposure pathway is expected to be an overestimate for most or all real individuals. The target individuals (assessment endpoint), exposure assumptions, and parameter values were chosen with this purpose in mind. The following sections describe the assessment endpoint, the contaminants and exposure pathways included in the analysis, the estimation of contaminant concentrations in water and sediment, and the selection of the decision guide. The equations used for screening-level estimation of risk are provided in Appendix 3A, and the estimated radionuclide concentrations in water and sediment and the various parameter values are provided in Appendix 3B. The results of the screening analysis--the screening indices for each radionuclide and exposure pathway and the comparison of the screening indices with the decision guide--are presented in summary form in Section 3.2.

3.1.1 Assessment Endpoint

Clinch River Mile (CRM) 20.8 near Jones Island (where White Oak Creek empties into the Clinch River) was chosen as the nearest location where potential exposures could have occurred to individuals living outside the Oak Ridge Reservation. Jones Island is the closest site to the mouth of White Oak Creek with documented human exposure. Beginning in October 1962 and ending in June 1963, the river channel adjacent to Jones and Grubb Islands was dredged to improve the channel for river transportation, and the spoils were placed on both islands. The dredged material was deposited at the head of Jones Island and, as the operation progressed, on the north side of the island (Morton, 1965). An individual at this site could

have received the maximum exposure through all pathways considered in this screening analysis because (1) this is the site nearest White Oak Creek with realistic potential for off-site exposure, and (2) higher water and sediment concentrations would be present here than downstream.

For all radionuclides with the exception of ^{131}I, the target individual for this analysis is defined as an adult who was exposed continuously from 1944 to 1991. Because the risk from ^{131}I is mainly associated with exposures in childhood, a child is used as the target individual of the screening assessment for ^{131}I. Parameter values (e.g., ingestion rates) were selected that are not expected to lead to an underestimate of risk to any real person (see Section 3.1.4).

3.1.2 Contaminants to be Screened

Available evidence indicates that the following radionuclides have been released to the Clinch River: ^{241}Am, ^{140}Ba, ^{144}Ce, ^{60}Co, ^{137}Cs, ^{154}Eu, ^{3}H, ^{131}I, ^{140}La, ^{95}Nb, ^{147}Nd, ^{32}P, ^{147}Pm, ^{143}Pr, $^{239/240}Pu$, ^{106}Ru, ^{151}Sm, ^{89}Sr, ^{90}Sr, ^{232}Th, ^{235}U, ^{238}U, ^{91}Y, and ^{95}Zr. The screening analysis was based on preliminary (February 1996) estimates of the amounts of these radionuclides released to the Clinch River. (The release estimates described in Section 5 were completed in late 1997 and reflect considerably more detailed study.) Preliminary release estimates for ^{232}Th, ^{235}U, ^{238}U, and the transuranium elements (TRUs; ^{241}Am and $^{239/240}Pu$) and for seven trivalent rare earths (TRE; ^{154}Eu, ^{140}La, ^{147}Nd, ^{147}Pm, ^{143}Pr, ^{151}Sm, and ^{91}Y) did not include isotopic breakdowns of these releases. Therefore, for screening purposes, the values labeled "TRU" (including ^{232}Th, ^{235}U, ^{238}U, ^{241}Am, and $^{239/240}Pu$) or "TRE" were used for each of the corresponding isotopes. In other words, for each isotope in turn, the TRU or TRE release estimate was conservatively assumed to consist entirely of the one isotope. A more detailed isotopic breakdown was later used for the screening estimates of ^{154}Eu and ^{140}La.

3.1.3 Estimation of Water and Sediment Concentrations

Preliminary estimates of the amounts of each radionuclide released from White Oak Lake each year were obtained from documentation of reported releases (e.g., Ohnesorge, 1986). Information from interviews (e.g., Harold Abee and Woody Cottrell) and literature searches indicated that the historic monitoring of radionuclide releases from White Oak Lake was inadequate and that documentation was fragmented. For example, in the early years, the flow rates across White Oak Dam were not measured during high flows, when the greatest releases occurred. Based on the interviews and on sediment measurements made in the White Oak Creek embayment below White Oak Dam, the highest realistic values for the releases over White Oak Dam were estimated to have been as much as a factor of 3 greater than the reported release estimates. As a result, the values used in this analysis for the amount of each radionuclide released over White Oak Dam in a given year were a factor of 3 greater than the total release estimate for that year documented from monitoring practices. These release estimates were intended to produce conservative estimates of water and sediment concentrations at the location of concern; in other words, they were not expected to lead to underestimates of exposure, but also were not thought to be unrealistically high.

For the TRUs and TREs, the upper-bound estimate of the release for each group was used for each radionuclide in the group. Since initial screening estimates, using very conservative assumptions, indicated that ^{154}Eu and ^{140}La could have posed a problem for the shoreline pathway, the component of the total TREs that was actually ^{154}Eu and ^{140}La was determined for the results presented here.

The upper-bound release estimates for each contaminant for each year are tabulated in Appendix 3B. These values were used as starting points for modeling of water and sediment concentrations for each year at CRM 20.8; the modeled concentrations of radionuclides in water and sediment for each year at CRM 20.8 (Appendix 3B) were used in the actual screening calculations. The models and assumptions used to produce these concentrations are described in Sections 3.1.3.1 through 3.1.3.3. Radionuclide-specific parameter values [e.g., radioactive decay constants, screening-level bioaccumulation factors, and screening-level distribution coefficients (K_d values)] are summarized in Appendix 3B.

3.1.3.1 Models Used for Estimation of Annual Average Radionuclide Concentrations in Water and Sediment

The primary model utilized in the present screening analysis was a modified version of MICHRIV (Delos et al., 1984); this model was modified to incorporate sediment input parameters obtained from an additional model, HEC-6-R (U.S. Army Corps, 1993). MICHRIV was selected as the primary screening tool because it describes both the physical and the chemical processes that determine radionuclide transport and fate in the sediment bed as well as in the water column. The principle calculations of the model were recorded in a spreadsheet (hereafter referred to as MICHRIV/SS). The disadvantage of MICHRIV is its inability to simulate sediment dynamics for multiple years. This limitation was overcome by using sediment data generated for the study area by a more sophisticated model (HEC-6-R) as an input variable and summing the net sediment flux (either deposition or scour) and its sorbed radionuclide concentration from year to year for the period 1944-1991. The customized MICHRIV/SS model was used to calculate water and sediment concentrations at CRM 20.8 for all radionuclides (tabulated in Appendix 3B).

3.1.3.2 Assumptions for Estimation of Water Concentrations

Concentrations of radionuclides in water at CRM 20.8 (Jones Island) were calculated based on the assumption of complete mixing in the water column. This assumption should not lead to an underestimation of actual water concentrations because of the influence of thermal stratification, which occurs when the warmer White Oak Creek water flows on top of the colder Clinch River water. If the radioactive release did reach the island, for example, under low flow conditions, the contaminant would be on top of the colder water; any withdrawal of water or sediment is most likely to have come from the river bottom, which would have a lower level of contamination. Assumption of complete mixing thus results in conservative estimates of deep water and sediment concentrations.

A second assumption used was that no filtration of the water took place and that no process, other than dilution, reduced the radionuclide concentration in water. In addition, the distribution coefficient (K_d) that describes the partitioning of the total radionuclide concentration into the adsorbed fraction (on particulates) and the dissolved fraction was set to 1 L kg^{-1}. This means that for the calculation of water concentrations, it

was conservatively assumed that radionuclides were not removed from the water column or deposited into the sediment bed.

3.1.3.3 Assumptions for Estimation of Sediment Concentrations

For estimating sediment concentrations, it was assumed that radionuclides were dispersed into the Clinch River at the time they went over White Oak dam. This is a reasonable assumption for radionuclides that stay in the water column (i.e., dissolved). It is a conservative assumption for radionuclides that are adsorbed to sediment because they may remain trapped in White Oak Creek embayment, sometimes for years, to be washed out during storm events. For example, the scouring of the embayment during 1956-57 is well documented (Blaylock et al., 1993). The assumption of immediate dispersion into the Clinch is conservative for estimation of sediment concentrations because radioactive decay that occurs in White Oak Creek embayment is thus discounted (Browder, 1959).

For calculation of sediment concentrations, the radionuclides were assumed to be evenly distributed throughout the study area. This conservative approach is likely to overestimate radionuclide deposition in the lateral direction (i.e., along the shoreline where human exposure is more likely to occur) because actual deposition occurs primarily in the deep river channel. In addition, the screening level distribution coefficient (i.e., the K_d value) was set to the maximum reported value for each radionuclide. This means that for calculation of sediment concentrations, the maximum amount was conservatively assumed to have been sorbed onto particulate matter. In addition, it was conservatively assumed that no loss of activity occurred due to deep burial in the sediment bed.

3.1.4 Exposure Pathways Included in the Analysis

The following scenarios were evaluated in the conservative screening analysis: ingestion of drinking water, ingestion of fish, swimming (including immersion and inadvertent ingestion of river water), external exposure to shoreline sediments, dredging (including external exposure, ingestion of beef, ingestion of milk, and ingestion of vegetables), and irrigation (including ingestion of contaminated peaches). These pathways were considered important routes of potential exposure because evidence suggested that they all represented opportunities for actual historical exposures of the public. Each pathway is discussed separately below; all exposure parameters with rationales for their selection are summarized in Appendix 3B. The discussions in this section are based on the information available to the project team in February 1996 and do not necessarily reflect the detailed studies described in later sections of this report.

3.1.4.1 Ingestion of Drinking Water

Although drinking water intakes are located on the Clinch River downstream from White Oak Dam, there has been no documented drinking water usage near Jones Island. Because higher water concentrations are expected nearer Jones Island than at the downstream water intakes, the screening analysis for this pathway was based on very conservative estimates of water concentrations. The total radionuclide screening indices for each radionuclide for the drinking water pathway were obtained by summing the estimated screening indices of excess lifetime cancer risk calculated for annual exposures from 1944 through 1991. This

represents a total effective exposure duration of 48 years. In addition, the screening indices were determined by assuming that the maximally exposed individual consumed 2.2 L d^{-1} of water, half of which was drawn from the river (unfiltered). The slope factors (SFs; excess lifetime risk of cancer incidence per Bq) were obtained from the Health Effects Assessment Summary Tables (HEAST; USEPA, 1995), with the exception of ^{131}I; the values are provided in Appendix 3B. For the screening-level estimation of the risk from ^{131}I, an ingestion slope factor was selected that represents a screening-level risk of excess thyroid cancer incidence for a child (the most at-risk individual; Apostoaei et al., 1995).

3.1.4.2 Ingestion of Fish

The ingestion of fish is thought to have provided a dominant route of exposure to the public. Evidence indicates that fishermen utilized a wide portion of the Clinch River and the Watts Bar Reservoir and that commercial fishing took place in the Watts Bar Reservoir and to a limited extent on the lower end of the Clinch River, which converges with the Tennessee River and is part of the Reservoir (Martin et al., 1964).

To provide a conservative estimate of the concentration of each radionuclide in fish, the upper-bound (screening-level) water concentrations were multiplied by a conservative (screening-level) bioaccumulation factor (IAEA, 1994). The target individual was assumed to have eaten an average of 30 g d^{-1}, 80% of which was assumed to be from the Clinch River. The same ingestion slope factors were used for fish ingestion as for ingestion of drinking water. The total radionuclide screening indices for each radionuclide for this pathway were obtained by summing the estimated screening indices of excess lifetime cancer risk calculated for annual exposures from 1944 through 1991.

3.1.4.3 External Exposure to Shoreline Sediment

Fishermen (recreational or commercial), swimmers, campers, and hikers could have been exposed to contaminated sediment along the shoreline of the Clinch River. For determination of screening indices for this pathway, two scenarios were used so that the effects of shielding by the water could be considered: (1) exposure during the part of the year when the shoreline is mostly covered (shielded) by the river water, and (2) exposure during the part of the year when the shoreline sediment is not covered. During the winter and spring, the water level is very low relative to the summer and fall levels because of drawdown of the Watts Bar reservoir for flood control purposes. In the summer and fall, the water covers most of the shoreline sediment, but in the winter and spring, the shoreline sediment is exposed. The highest potential rate of exposure thus would normally occur in the winter and spring. A person can fish standing on the sediment in the winter and spring but must fish from the bank in the summer and fall. However, even though the highest rates of exposure would have occurred in the winter and spring, people are not fishing or hiking as often during these times of year. The rationales for the values selected for exposure frequency to the shoreline sediment are presented in Appendix 3B. Dose factors for external exposure to surface-level sediment were obtained from the U.S. Environmental Protection Agency's Federal Guidance Report No. 12 (USEPA, 1993) and are listed for each radionuclide in Appendix 3B.

3.1.4.4 Swimming (Immersion and Inadvertent Consumption of Water)

The screening indices for the swimming scenario were estimated by summing the screening index based on excess lifetime cancer risk from external exposure (from immersion) and the screening index based on excess lifetime cancer risk from inadvertent ingestion of Clinch River water while swimming. For this scenario, it was conservatively assumed that a person swam in the river downstream from White Oak Dam approximately 4 hours per day for about 4 months of the year. In addition, inadvertent ingestion of river water was assumed to have been about 0.05 L hr^{-1} spent swimming. Dose factors for external exposure from immersion in water were obtained from the U.S. Environmental Protection Agency's Federal Guidance Report No. 12 (USEPA, 1993), and radionuclide SFs for ingestion were obtained as described in Section 3.1.4.1. These values are summarized in Appendix 3B.

3.1.4.5 Dredging Scenario

This scenario, which includes several exposure pathways, was analyzed to estimate the effects of historical dredging of deep sediments that occurred near Jones Island. A navigation lock was constructed as part of Melton Hill Dam to improve navigation along the Clinch River (Morton, 1965). To further facilitate navigational use, channel improvements along the Clinch River were also carried out. The channel improvements involved the removal of bottom sediments via either a hydraulic dredge or a dipper dredge. The dredged material was deposited onto nearby land in spoil banks or beds. The Jones Island area was dredged in May 1963; Jones Island was used as the depository for material dredged between CRM 19.5 and 20.8. All dredging activities were completed in June 1963 (Morton, 1965).

The Jones Island dredging was initiated at the head of the island. As the dredging operation moved along the island, the deposition of spoils also moved down the island. The spoils were deposited in ponds that had been excavated on the north side of the island and at the lower end of the island. Even though most documentation suggests that the spoils were placed primarily on one end of the island, it was assumed for this screening level analysis that the dredged spoils were placed uniformly across the entire island. This assumption leads to very conservative estimates of exposure for the purposes of this screening analysis. In addition, it was assumed that dredging occurred once; therefore, the screening indices were estimated by integrating over 30 years (1963 to present day).

The following pathways were included for the dredging scenario: (1) external exposure to humans; (2) ingestion of vegetables grown on Jones Island; (3) ingestion of beef raised on Jones Island; and (4) ingestion of milk from cows kept on Jones Island. Evidence indicates that beef cattle were raised on Jones Island and that the cattle used river water directly as their source of drinking water. Therefore, the exposure estimate for beef and dairy cows kept on Jones Island had three different consumption inputs: contaminated soil, pasture grown on contaminated soil, and contaminated drinking water. The total radionuclide screening indices for ingestion of milk and beef were estimated by summing the integrated screening indices for the dredged sediment pathways (ingestion of contaminated soil and contaminated pasture by the cows) and the summed screening indices obtained for annual screening-level estimates of exposures from consumption of milk or beef from cows drinking contaminated river water.

3.1.4.6 Irrigation Scenario

During the period 1950-1955, irrigation with water from the Tennessee River was significantly increased (Morton, 1965). However, according to census information, Roane County, which has farming tracts on both the Clinch and Tennessee Rivers, had only 6 farms irrigating 39 acres of land in 1954, and 5 farms irrigating 13 acres in 1959 (TVA, 1963). In 1965, crop irrigation along the Clinch River was nonexistent, and there was limited irrigation along the Tennessee River downstream from Oak Ridge (Zirkle, personal communication with C. Lewis, 1996). Since some irrigation with Clinch River water was conducted near the Kingston Steam Plant, the irrigation pathway was examined in this conservative screening analysis. The rationale for each of the parameters used in this scenario is included in Appendix 3B.

3.1.5 Specific-Activity Calculation for Tritium

The screening calculation for tritium (^{3}H) was based on a specific-activity approach (IAEA, in preparation) to account for the variety of chemical compounds and corresponding potential exposure pathways that can include tritium. This model is expected to give a conservative dose estimate based on an assumption of complete equilibrium between the exposed individual and the specific activity of ^{3}H in the environment. The basic equations used for the calculation are

$$D_T^{max} = [(C_A^{max} \cdot f_A) + (C_W^{max} \cdot f_W)]\, g$$

and

$$SLRE_{H\text{-}3} = D_T^{max} \cdot RCF, \text{ summed over all years}$$

where

D_T^{max} is the dose rate (Sv y^{-1}) for ^{3}H to the whole body of a maximally exposed individual;

C_A^{max} is the steady-state concentration of ^{3}H in atmospheric water vapor (Bq L^{-1}) at the location of interest resulting from atmospheric releases (a function of the concentration of ^{3}H in air and the absolute humidity of the atmosphere at the specified location);

f_A is the fraction of total water intake derived from atmospheric water vapor at the location of interest (through all pathways, including inhalation and ingestion of foods produced at that location);

C_W^{max} is the steady-state concentration of ^{3}H in water (Bq L^{-1}) resulting from releases to the aquatic environment;

f_W is the fraction of total water intake derived from the contaminated water source at the location of interest (through all pathways, including ingestion of drinking water, foods irrigated with contaminated water, or products from animals that drank contaminated water);

g is the dose rate conversion factor (Sv y^{-1} per Bq L^{-1} of human body water content);

$SLRE_{H\text{-}3}$ is the screening-level risk estimate for exposure to tritium; and

RCF is a risk conversion factor (risk Sv^{-1}).

The present analysis dealt only with aquatic releases of tritium; therefore C_A^{max} was considered to be 0 and the value of f_A was not important. The yearly values for C_W^{max} are listed in Table 3B-5; f_W was conservatively set at 0.75, corresponding approximately to the assumptions used for other radionuclides in this analysis in terms of the consumption of river water or of food products affected by the river water. The dose rate factor g for ^3H is 2.6×10^{-8} Sv y^{-1} per Bq L^{-1} (NCRP, 1979). A value of 0.073 Sv^{-1} (ICRP, 1990) was used for the risk conversion factor (Table 3B-1).

Based on these equations and parameter values, a screening index of 8.7×10^{-6} was obtained for ^3H. This screening index represents a conservative estimate of the risk from all exposure pathways originating from the contaminated river water; values for ^3H corresponding to the pathway-specific screening indices calculated for the other radionuclides in this analysis would be lower.

3.1.6 Use of a Decision Guide

A decision guide of 10^{-5} lifetime risk of excess cancer incidence was used in this screening analysis. This level is a factor of 10 lower than the ORHASP's current decision guide of 10^{-4} lifetime cancer risk (Thiessen et al., 1996). A value of 10^{-5} was used because each radionuclide was compared to the decision guide independently for each exposure pathway. Using the more conservative decision guide for the screening analysis results in high confidence that the radionuclides assigned low priority for a pathway do not in fact contribute significantly to the overall dose or risk for that pathway.

For this screening analysis, each screening index was compared to the 10^{-5} decision guide as follows:

- If the screening index for an individual radionuclide and pathway was clearly below the decision guide, further study of the contaminant for that pathway was deferred until such time as resources permit further analysis. The logic is as follows: If the maximally exposed target individual has a low screening index for a contaminant (i.e., the screening estimate of risk for that contaminant is below the decision guide), then the true but unknown risk to members of the general population is expected to be even lower. Continued expenditure of time and resources for evaluation of that particular contaminant for a given exposure pathway is not justified as long as there are more important situations to be studied.

- If the screening index for a given radionuclide and pathway was at or above the decision guide, the radionuclide was evaluated in more detail for that pathway in the next phase of Task 4 (later sections of this report).

3.2 Results of the Screening Analysis

The screening calculations were based on generic equations for calculation of dose and risk (Appendix 3A); the effects of radioactive decay and of exposure and release duration were included. The calculations included all pathways expected to be significant and for which available evidence (at the time of the screening analysis) indicated that the pathway occurred or could have occurred historically.

Parameter values for contaminant-independent parameters, with rationales for their selection, are provided in Appendix 3B (Table 3B.1). Parameters which depend on the specific contaminant or the site-specific situation are also presented in Appendix 3B (Tables 3B.2, 3B.3, 3B.4, and 3B.5). These contaminant-specific parameters include toxicity values, radionuclide decay constants, transfer factors, bioaccumulation factors, and distribution coefficients. The last three parameter types listed consist of screening-level (i.e., conservative) values, based on the information available at the time of the analysis.

The results of the conservative screening analysis are presented in Table 3.1. These screening values represent conservative estimates of excess lifetime risk of cancer incidence from an exposure duration equal to the number of years of historical releases. The contaminants and pathways with a screening index above 10^{-5} have been analyzed in more detail in the remaining sections of the Task 4 report. The radionuclides that were analyzed further are summarized in Table 3.2.

Based on the results of the conservative screening analysis, the swimming scenario was given a low priority for further study. Furthermore, because the conservative screening analysis of the irrigation scenario did not produce a screening value above 1×10^{-5} lifetime risk and the only produce irrigated was peaches (for approximately 3 months per year), this pathway was not considered for further study. In addition, sixteen radionuclides were assigned low priority for further effort in this task: ^{32}P, ^{154}Eu, ^{140}La, ^{140}Ba, ^{89}Sr, ^{91}Y, ^{147}Nd, ^{143}Pr, ^{3}H, ^{235}U, ^{238}U, $^{239/240}$Pu, ^{232}Th, ^{241}Am, ^{147}Pm, and ^{151}Sm. Of the remaining eight radionuclides, ^{137}Cs, ^{60}Co, ^{106}Ru, and ^{90}Sr were expected to be the most important, with ^{131}I, ^{144}Ce, ^{95}Zr, and ^{95}Nb providing smaller contributions to doses and risks.

All pathways that involved direct ingestion of contaminated food or water resulted in a screening index for ^{131}I above the 10^{-5} decision guide applied in this analysis. The equations for most of the radionuclides identified in the milk and beef ingestion pathway did not contain a term to account for the decay of radionuclides between harvest and consumption of milk or beef, resulting in further conservatism in the estimate. However, a decay term was used for ^{131}I for the beef pathway. The screening index for ^{131}I from ingestion of beef was below the 10^{-5} decision guide. However, ^{131}I exposures from other pathways are examined in more detail in later sections of this report.

Table 3.1 Conservative screening indices for radionuclides in the Clinch River[1].

Isotope	Drinking Water	Fish Ingestion	External: Shoreline	Swimming	External: Dredged Sediment	Ingestion of Beef	Ingestion of Milk	Ingestion of Vegetables	Irrigation
Cs-137	**9.2 E-06** [2]	**4.0 E-04**	**8.0 E-03**	7.6 E-07	**1.6 E-03**	**5.9 E-03**	**5.7 E-03**	**5.6 E-04**	3.2 E-08
Ru-106	**7.7 E-05**	**1.7 E-05**	**1.1 E-03**	5.2 E-06	**4.5 E-05**	**1.6 E-04**	4.4 E-07	**5.8 E-05**	1.2 E-08
Sr-90	**2.5 E-05**	**3.3 E-05**	**7.1 E-05**	1.5 E-06	**9.8 E-06**	**1.7 E-02**	**2.5 E-02**	**6.4 E-03**	5.1 E-07
Co-60	2.8 E-06	**1.9 E-05**	**6.0 E-03**	1.7 E-07	**8.5 E-04**	**1.1 E-03**	**7.6 E-04**	**7.5 E-05**	6.2 E-09
Ce-144	4.2 E-06	2.7 E-06	**2.1 E-05**	2.6 E-07	7.2 E-08	1.1 E-08	7.4 E-08	3.2 E-07	2.2 E-09
Zr-95	8.1 E-07	5.3 E-06	**1.8 E-04**	4.3 E-07	5.1 E-09	8.8 E-11	2.7 E-10	2.1 E-12	3.1 E-12
Nb-95	4.2 E-07	2.7 E-06	**5.1 E-05**	2.0 E-07	3.1 E-09	1.4 E-11	9.1 E-11	1.4 E-11	3.7 E-12
I-131	**4.1 E-05**	6.7 E-06	7.2 E-08	4.1 E-06	3.2 E-12	6.0 E-07	**3.8 E-05**	1.1 E-11	9.3 E-10
U-235	1.5 E-07	3.2 E-08	5.0 E-06	9.4 E-09	7.8 E-07	2.8 E-07	2.7 E-07	4.6 E-07	1.8 E-10
U-238	1.3 E-07	2.9 E-08	8.4 E-07	8.0 E-09	1.4 E-07	2.5 E-07	2.4 E-07	4.2 E-07	1.6 E-10
Pu-239/24	9.8 E-07	6.4 E-07	1.4 E-07	5.9 E-08	1.5 E-09	3.8 E-07	2.8 E-08	3.1 E-06	2.4 E-10
Th-232	1.0 E-07	2.2 E-07	9.2 E-08	6.1 E-09	2.7 E-09	2.0 E-08	4.8 E-09	1.6 E-07	1.2 E-11
Am-241	1.0 E-07	6.7 E-08	3.8 E-06	6.2 E-09	2.0 E-07	1.7 E-08	1.6 E-08	2.8 E-07	2.5 E-11
Eu-154	4.9 E-06	5.3 E-06	3.6 E-08	1.1 E-06	5.1 E-09	1.3 E-06	1.7 E-07	1.0 E-06	4.4 E-10
La-140	4.9 E-06	2.7 E-06	1.0 E-06	1.8 E-06	2.0 E-09	1.1 E-07	1.6 E-08	7.2 E-12	3.9 E-13
Pm-147	7.4 E-07	4.8 E-07	2.6 E-08	4.4 E-08	1.1 E-11	1.7 E-08	2.8 E-09	6.0 E-10	3.6 E-11
Sm-151	2.3 E-07	1.5 E-06	1.3 E-07	1.4 E-08	3.8 E-10	9.0 E-07	1.2 E-07	7.5 E-07	2.7 E-11
Sr-89	1.5 E-08	1.9 E-08	1.2 E-11	8.8 E-10	1.1 E-13	1.4 E-09	2.4 E-09	3.4 E-11	0.0 E+00
Ba-140	8.6 E-07	9.4 E-08	5.6 E-07	2.8 E-07	0.0 E+00	1.9 E-09	2.3 E-08	0.0 E+00	5.4 E-12
P-32	7.8 E-08	3.8 E-06	2.3 E-12	4.7 E-09	6.9 E-16	4.2 E-08	6.8 E-08	3.3 E-13	1.6 E-13
Y-91	7.0 E-06	4.6 E-06	3.5 E-07	4.2 E-07	9.3 E-11	7.6 E-08	2.3 E-08	1.1 E-10	2.9 E-11
Pr-143	3.5 E-06	2.3 E-06	9.6 E-09	2.1 E-07	1.5 E-12	7.6 E-08	1.1 E-08	8.3 E-12	0.0 E+00
Nd-147	3.1 E-06	2.0 E-06	1.6 E-06	2.7 E-07	3.6 E-10	6.8 E-08	1.0 E-08	6.0 E-12	0.0 E+00

1 The screening index for tritium (not included in this table) was calculated as described in Section 3.1.5.
2 **Bold** values represent radionuclides for each pathway that were carried into the next iteration of analysis in Task 4.

Table 3.2 Summary of Radionuclides and Pathways Identified for Further Analysis.

Pathway	Isotopes
Drinking Water	Cs-137, Ru-106, Sr-90, I-131
Fish Ingestion	Cs-137, Ru-106, Sr-90, Co-60
External: Shoreline Sediment	Cs-137, Ru-106, Sr-90, Co-60, Ce-144, Zr-95, Nb-95
Swimming	None
Dredging Pathways:	
External	Cs-137, Ru-106, Sr-90, Co-60
Ingestion of Beef	Cs-137, Ru-106, Sr-90, Co-60
Ingestion of Milk	Cs-137, Sr-90, Co-60, I-131
Ingestion of Vegetables	Cs-137, Ru-106, Sr-90, Co-60
Irrigation	None

Finally, the two TREs (^{154}Eu and ^{140}La) that were of initial concern because of the conservative treatment of their source term were given a lower priority for further consideration when the actual isotopic breakdown of the TRE release data was used.

A screening index of 8.7×10^{-6} was obtained for ^{3}H (Section 3.1.5). This screening index represents a conservative estimate of the risk from all exposure pathways originating from the contaminated river water; screening values for ^{3}H corresponding individual pathways would be lower. Therefore, tritium was assigned a low priority for further study.

3.3 Summary of the Screening Analysis

Twenty-four radionuclides released into the Clinch River from the Oak Ridge National Laboratory from 1944-1991 were considered in a preliminary screening analysis conducted in 1996 to determine whether a more in-depth evaluation would be required in later phases of Task 4. To focus time and resources on those radionuclide contaminants that are most likely to have been important in terms of dose or risk to off-site individuals, a conservative screening evaluation of each individual radionuclide and exposure pathway was conducted.

To identify contaminants and pathways for which the risk was clearly below a minimum level of concern, conservative assumptions were used for all aspects of the screening calculations. Because each radionuclide was evaluated individually by pathway, the results of the screening calculations were compared to a risk-based decision guide of 10^{-5} (a factor of 10 below the ORHASP recommended value), and priority designations were developed for each contaminant and pathway.

As a result of this conservative screening analysis, sixteen radionuclides were assigned a low priority for further analysis. In addition, the swimming and irrigation scenarios were designated as low priority exposure pathways. Eight radionuclides (^{137}Cs, ^{160}Ru, ^{90}Sr, ^{131}I, ^{60}Co, ^{144}Ce, ^{95}Zr, and ^{95}Nb) were investigated in more detail (later sections of this report). The exposure pathways of primary concern warranting further analysis were as follows: drinking water, fish ingestion, external exposure from sediments, ingestion of milk, and ingestion of meat. These more in-depth evaluations were conducted in 1996 and 1997 and are documented in the remaining sections of this Task 4 report.

3.4 References

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4.0 OVERALL APPROACH

Task 4 of the Oak Ridge Dose Reconstruction consists of an evaluation of radionuclide releases from White Oak Lake into the Clinch River. Estimated and measured concentrations of radionuclides in water and shoreline sediment were used to calculate total excess lifetime risks of cancer incidence for defined target individuals in specific locations for each of five exposure pathways.

Eight potential exposure pathways were considered in the screening phase of Task 4 (Hammonds et al., 1997; Section 3 of this report): ingestion of fish, swimming, ingestion of drinking water, ingestion of milk, ingestion of meat, external exposure from shoreline sediment, external exposure from dredged sediment, and ingestion of produce or other crops grown on land contaminated by dredging or irrigation. The screening analysis used a risk criterion of one chance in one-hundred thousand of an adverse health outcome (10^{-5}) to assign low priority for further study. Following the screening analysis and a demographic survey of the affected area (Section 7), four pathways were excluded from the full risk assessment for all locations along the Clinch River because the screening results were below the decision criterion in the conservative screening analysis or the pathways were not relevant to the actual land use along the Clinch River. The excluded pathways are swimming, external exposure from dredged sediments, ingestion of crops grown on land contaminated by dredging, and ingestion of crops grown on land contaminated by irrigation. For each of the remaining five pathways, appropriate reference individuals were defined for specified locations.

The study area was defined as the region along the Clinch River from the Jones Island Area [Clinch River Mile (CRM) 20.5; just downstream from White Oak Creek, the source of the radioactive contaminants from the Oak Ridge Reservation] to the confluence of the Clinch River with the Tennessee River (CRM 0.0). Five specific locations were identified for inclusion in the analysis. These locations are the Jones Island Area (referred to as CRM 20.5 on tables and figures), the Grassy Creek Area (referred to as CRM 14 on tables and figures), the K-25 Area (also referred to as CRM 14 on tables and figures), the Kingston Steam Plant Area (referred to as CRM 3.5 on tables and figures), and the City of Kingston (referred to as CRM 0.0 on tables and figures). Although Grassy Creek and the K-25 area are defined as distinctly different locations, the only difference in terms of the presence of exposure pathways is that there is public use of the river as a drinking water source at the K-25 Water Intake. However, because the predicted water concentrations in the Clinch River are not different for these locations, they are referred to as part of a common reach of the river for purposes of the risk assessment. Information on demography and land use was utilized to evaluate each location for the possibility that people might have been exposed through any of the five pathways (Section 7).

This section of the report provides the general methodology used in each component of the dose reconstruction process. Section 4.1 describes the basic conceptual model used to estimate the risks from the radionuclides of concern. Section 4.2 contains an introduction to the methods used to evaluate each component of the dose and risk estimation. Finally, Section 4.3 provides a brief overview of the methods used to perform the uncertainty and sensitivity analyses.

4.1 Conceptual Model

The excess lifetime risk (*ELR*) of cancer from exposure to a radionuclide of concern is defined as the probability that an individual exposed to a single dose D_i in a given year (*i*) will acquire a radiation-induced cancer over that individual's lifetime. In this study, the excess lifetime risk of cancer was estimated for each organ of interest (Section 13). The excess lifetime risk of cancer was then calculated for several reference individuals exposed as adults (with the exception of ^{131}I, for which the reference individual was considered to be exposed as a child) to specified radionuclides released from the X-10 facility (Section 5). The excess lifetime risk of cancer incidence during year "*i*" (ELR_i) was calculated using the equation

$$ELR_i = RF_i \cdot D_i \qquad (4.1)$$

where

RF_i = Organ-specific Risk Factor, excess lifetime risk of cancer to an organ per unit dose from exposure during year "*i*" [Sv^{-1}]

and

D_i = Dose delivered to the organ from an exposure in year "*i*" [Sv].

If the individual was exposed for more than 1 year, the total excess lifetime organ-specific risk ($TELR_j$) of cancer was given by

$$TELR_j = \sum_{i=1}^{N} RF_i \cdot D_i \qquad (4.2)$$

where *N* is the number of exposure years.

The historical releases of radionuclides from the Oak Ridge National Laboratory relevant to this study occurred for 48 years, from 1944 to 1991.

To obtain the total excess lifetime risk of cancer from multiple exposures to all organs, the following model was used:

$$TELR = \sum_{j=1}^{N} TELR_j \qquad (4.3)$$

where *N* is the number of organs for which a radiological dose and dose response function was estimated.

A process-based approach was used to estimate the total excess lifetime risk (*TELR*). The excess lifetime risk (*ELR*) from an annual exposure at a specified location was calculated as a function of several independent components:

- the source term, or the yearly release of a radionuclide to the river;
- the aquatic transport, or the resulting annual average radionuclide concentrations in water and shoreline sediment at specified locations;
- food-chain transfer from contaminated water to edible fish, milk, or meat;
- intake of contaminated water or food, or exposure to contaminated shoreline sediment;
- dose per unit intake or unit exposure; and
- risk per unit dose.

A detailed list of the underlying equations used for the assessment of all relevant pathways is given in Appendix 4A. These equations were used to estimate the intake of a contaminant (in Bq) by a given pathway from an initial contaminant concentration in water (internal exposure pathways) or the external exposure (Bq kg^{-1} y) from an initial contaminant concentration in shoreline sediment. Doses (Sv) were then calculated from the total intake or total exposure. Estimation of the radionuclide concentrations in water and sediment as functions of time and location is described briefly in Section 4.2 and in detail in Sections 5 and 6.

4.2 Description of Model Components

This section introduces the general methodology used to determine each component listed in the previous section. Details of the methodology described here are provided in later sections of this report.

4.2.1 Source term estimation

Source term estimation was used to quantify the amounts of each radionuclide of interest released in liquid effluents from White Oak Dam and subsequently to the Clinch River during the years 1944-1991. A detailed source term was developed for each of the following radionuclides: ^{60}Co, ^{90}Sr, ^{95}Nb, ^{95}Zr, ^{106}Ru, ^{131}I, ^{137}Cs, and ^{144}Ce.

The method used to estimate the source terms for these radionuclides involved several uncertain correction or bias factors on the reported annual release estimates. The sources of uncertainty varied over time due to changes in the operations of the facilities and due to changes in monitoring and analysis methods. Further details on the development of the detailed source terms are discussed in Section 5.

4.2.2 Estimation of radionuclide concentrations in surface water and shoreline sediments

Estimates of the actual radionuclide concentration in surface water and in shoreline sediments are central to reconstructing possible exposures. Several factors must be considered, such as the radioactive half-life, flow rate of the river system, effects of dilution or water volume, the form of particulate matter, the suspension of particulate matter, the resuspension of particulates, scouring events, etc. The estimation of water and sediment concentrations is discussed in Section 6.

Estimates of Water Concentrations

Estimates of water concentrations were obtained using a modified version of an aquatic transport model called HEC-6 (HEC-6-R; see Section 6). However, for the purposes of the dose reconstruction for Task 4, measured data were used whenever possible to estimate annual average water concentrations in the Clinch River. The uncertainty in the modeled water concentrations from HEC-6R was much higher than the uncertainty about annual averages obtained from the measured data, primarily because the model was not calibrated to specific water concentrations (calibration was performed on the basis of total sediment inventory). Because the concentration data themselves would be necessary to calibrate the model, it was considered best to use the concentration data directly when they were available. Another reason to prefer the use of measured data over modeled predictions is that the measurements of water concentrations are not affected by uncertainty in the estimates of releases of radionuclides over White Oak Dam.

For those years for which environmental data exist, the reported values were adjusted for the locations of interest and used with appropriate uncertainty factors. For those years for which environmental data do not exist, subjective confidence intervals based on the predictions from HEC-6-R were used.

Estimates of Sediment Concentrations

Estimates of shoreline sediment concentrations were also obtained using HEC-6-R (Section 6). A complicated model, HEC-6-R was used to track the sediment inventory in various reaches of the Clinch River. Various flood and scouring events were also included in the modeling effort.

The basic process of estimating sediment concentrations uses known information about water concentration, particulate type, suspension of particulate matter, and K_d values. The K_d value (distribution coefficient) of a radionuclide describes how well the radionuclide is absorbed on to sediment material. Some radionuclides such as ruthenium have a very low K_d, meaning that very little is absorbed and retained on the river sediments. Instead, the radionuclide remains primarily in the water, and higher exposures would result from the water carrying the ruthenium than from the sediments. Cesium-137 has a high K_d value, meaning that it is strongly absorbed by the sediments. As a result, continuing exposure could take place from both sediment and water, due to continuing releases to the water and to resuspension of contaminated sediment.

Monitoring data obtained in the early 1960s and in the 1990s were used to calibrate the model for shoreline sediment estimation. The primary calibration was based on sediment inventory during the 1960s. For the

shoreline sediment, all concentrations used in the risk estimation were based on model calculations, with appropriate uncertainty estimates.

4.2.3 Estimation of radionuclide concentrations in aquatic and terrestrial biota

Aquatic and terrestrial biota living in proximity to the Clinch River or the contaminated sediments had some possibility of accumulating radionuclides in their tissues. Accumulation of a radionuclide depends first on the bioavailibility of that individual radionuclide. For example, ^{137}Cs is highly bioavailable and subject to accumulation in biota, whereas ^{106}Ru is not. Cesium-137 is therefore of more concern for an ingestion pathway such as the consumption of fish, meat, or milk than is ^{106}Ru.

To estimate the risk from ingestion of fish, a bioconcentration factor (BCF) was used to predict the transfer of contaminants from water to the fish. A bioconcentration factor with an estimate of the associated uncertainty was developed for each radionuclide of concern. These BCFs were estimated from available environmental data (water concentrations and fish concentrations) and from published information. The detailed development of the BCFs used to estimate the transfer of radionuclides from river water to fish is provided in Section 8.

To estimate the risk from ingestion of milk and meat, radionuclide-specific coefficients were used to predict the transfer from river water to milk and meat. These transfer coefficients are described in Section 9.

4.2.4 Estimation of exposures to target individuals

For all locations in the study, exposure pathways of interest included fish ingestion and ingestion of milk and meat; other pathways of interest varied with location. For the Jones Island Reach (CRM 21.0 to CRM 17.0), the exposure pathways of interest are fish ingestion, external exposure from shoreline sediments, and ingestion of milk and meat. The Grassy Creek Reach (CRM 17.0 to CRM 14.0) exposure pathways include fish ingestion, external exposure to shoreline sediment, and ingestion of milk and meat, while the nearby K-25 reach (CRM 14.0 to CRM 5.0) includes the addition of the drinking water pathway. For the Kingston Steam Plant (CRM 5.0 to CRM 2.0), the important pathways were drinking water, fish ingestion, external exposure to shoreline sediments, and ingestion of milk and meat. Finally, exposures to residents in the City of Kingston (CRM 2.0 to CRM 0.0) are assumed to have occurred primarily from drinking water, fish ingestion, external exposure to shoreline sediments, and ingestion of milk and meat from livestock having direct access to the river as a source of drinking water.

Estimation of exposures to reference individuals identified in this study depends on the pathway involved, the specific characteristics of the individual pathways, and the size and type of the population affected. For the fish pathway, three reference individuals were described in terms of the amount of fish consumed. The water ingestion pathway has two reference individuals, an adult and a child. The actual amount of time the reference individual had access to contaminated drinking water was also taken into account. For example, children are not considered for certain areas such as the Grassy Creek/K-25 area and the Kingston Steam Plant, because these are industrial facilities and it is not likely that children obtained drinking water from these locations. On the other hand, adults and children were exposed via the Kingston City Water Supply.

The milk ingestion pathway also had multiple target individuals: children, who could have drunk different fractions of locally produced milk depending on whether they were at home or in school, and adults. Both the meat ingestion pathway and the external exposure pathway had adults as reference individuals. Details on the exposure scenarios and reference individuals are presented in Section 7.

Several special exposure scenarios were also considered in this study. These scenarios included a person who ate fish patties composed of whole fish as well as fish fillets, a person consuming a fish from White Oak Lake, an individual consuming a contaminated deer from the reservation, an individual consuming a contaminated waterfowl from the reservation, and an individual consuming a contaminated turtle from White Oak Lake. These special scenarios are presented in Section 14.

4.2.5 Estimation of dose conversion factors

For estimation of doses to the reference individuals, doses to specific organs per unit intake (dose conversion factors) were developed for the ingestion pathways based on internal dosimetry concepts and methods established by the International Commission on Radiological Protection (ICRP). Sources of uncertainty in the dose conversion factors for each radionuclide and organ of concern were identified, estimated, and propagated to determine the uncertainty in the ICRP methods.

In order to predict doses from the external pathway, an "external" dose conversion factor is required. To estimate the uncertainty in the external dose conversion factors, the published values were evaluated in terms of their applicability to the Clinch River system. Several modifying factors were determined and propagated to obtain estimates of uncertainty for the external dose conversion factors.

Further discussion of the methodology used to estimate the uncertainties in the organ-specific dose conversion factors for both external and internal exposures is provided in Sections 10 and 11, respectively.

4.2.6 Estimation of dose-response functions and total excess lifetime risks

The excess lifetime organ-specific risk per unit organ-specific dose is discussed in Section 12 as a function of the background incidence of organ-specific cancer incidence in the state of Tennessee. The effects of age and gender are also discussed in Section 12, along with the estimation of total risk for all cancers. Finally, the overall doses and the total excess lifetime risks of cancer incidence for the target individuals are summarized and presented in Section 13.

4.3 Uncertainty and Sensitivity Analysis

Each of the components listed in Section 4.2 contains parameters whose values are known only imperfectly. To quantify the present state of knowledge, subjective probability distributions were assigned using professional judgment after review of the literature, site-specific data, and consultation with outside experts (IAEA, 1989; NCRP, 1996). Whenever possible, distributions of values have been selected to reflect specific characteristics of the Oak Ridge site and of activities along the Clinch River during the time period 1944 through 1991. Subjective probability distributions for all parameters were propagated through the equations to give subjective probability distributions for the resulting doses and risks.

Monte Carlo simulation (Figure 4.1) was used for error propagation, employing the Latin Hypercube Sampling (Crystal Ball: Decisioneering, 1994) technique with a sample size of 400. Identification of the most important contributors to the uncertainty in dose and risk was performed through a sensitivity analysis. In this document, the sensitivity analysis expresses the relative contribution of the uncertainty of each input parameter to the uncertainty in each endpoint of the calculation. This analysis is based on a summation of the squares of the Spearman rank correlation coefficients[1] (between each input parameter and the endpoint of interest), normalized to 100% (Crystal Ball: Decisioneering, 1994). Once the important contributors to the overall uncertainty have been identified, priority can be assigned for the acquisition of additional information to refine the dose and risk estimates.

A specific feature of the sensitivity analysis performed in this work is the treatment of the time-dependent nature of the processes involved. The main endpoints of the calculations, namely the total excess lifetime risk (*TELR*) of cancer incidence and the total dose (*TD*) to selected organs, represent time-integrated quantities. Their relationship to the time-dependent parameters is complex and difficult to analyze. The solution chosen for the sensitivity analysis was to investigate the relationship between the uncertainty in the above-mentioned endpoints and the uncertainty in the time-averaged values for the time-dependent parameters.

Correlations between most parameters in the exposure equations were considered minimal and were not treated explicitly. Total fluid intake by children was considered in estimation of the separate ingestion rates for milk and drinking water. Correlation between exposure pathways in terms of dependence on the same contaminant concentrations in water for a given year and location was included explicitly; total doses and risks for all pathways at a given location were estimated using the same starting concentration of each contaminant for all pathways. In other words, for each value sampled from a distribution of contaminant concentrations in river water, all contributions to dose (i.e., consumption of fish, milk, beef, and water) were calculated from the same value; this process was repeated 400 times for each dose or risk estimate. Thus the resulting distribution for dose or risk accounted for the fact that all internal exposure pathways were based on the same water contamination. Correlation between contaminant concentrations in water and in shoreline sediment was not treated explicitly, but it is expected that the large uncertainty in the estimated concentrations in sediment will exceed any effect on total dose or risk from omitting this correlation.

[1]*Spearman rank correlation coefficients* are calculated between each input parameter and the output. Correlation coefficients provide a meaningful measure of the degree to which an input parameter and the result *change together*. A high correlation coefficient means that the input parameter has a significant impact on the result. A positive coefficient indicates that the result increases as the input parameter increases. For a negative coefficient the result decreases as the input parameter increases. The larger the absolute value of the correlation coefficient, the stronger the relationship (Crystal Ball: Decisioneering, 1994).

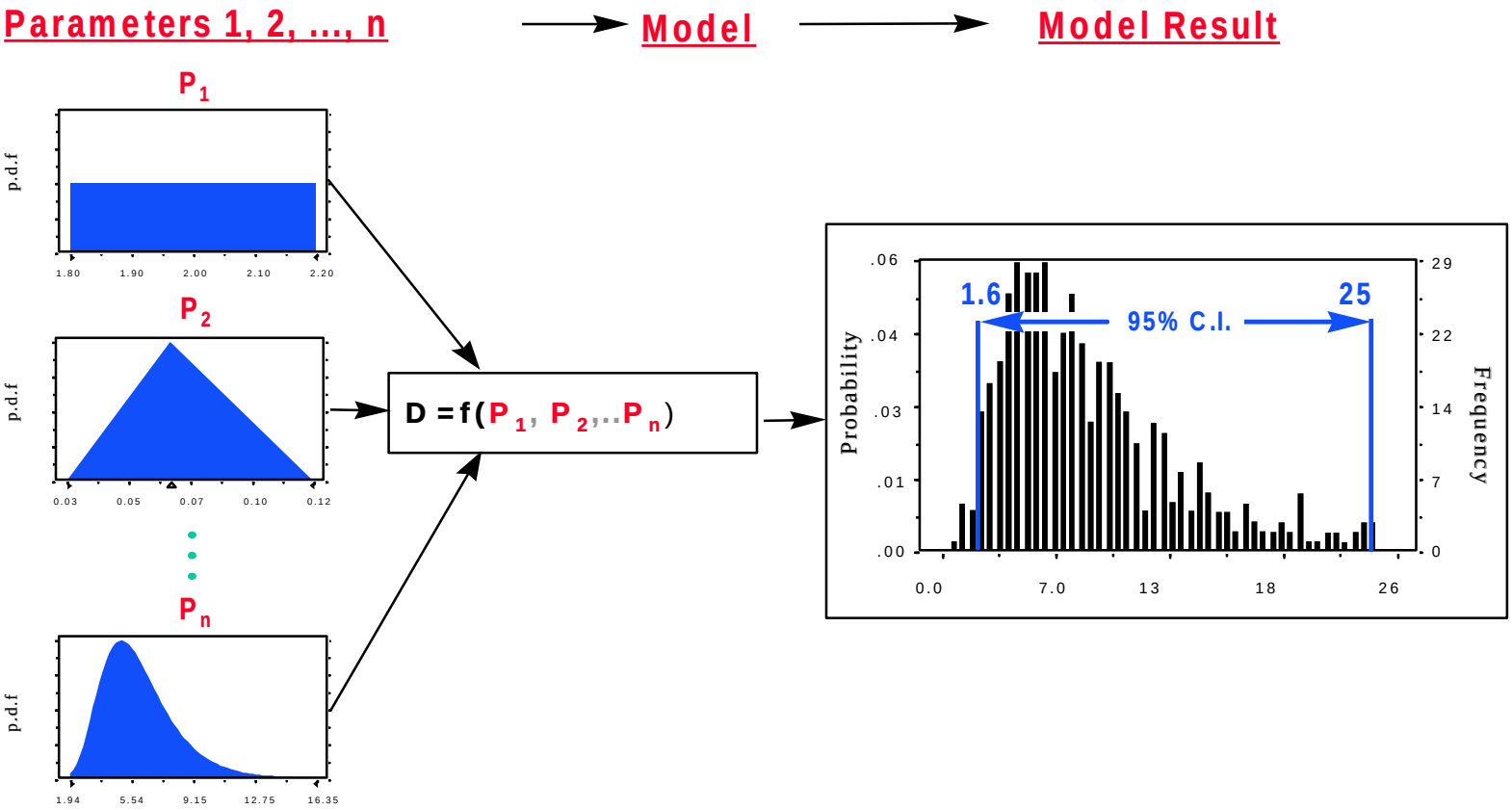

Figure 4.1 Monte-Carlo method for uncertainty propagation

4.4 References

Crystal Ball: Decisioneering. 1994. "Crystal Ball® Version 3.0, Forecasting and Risk Analysis for Spreadsheet Users" 1380 Lawrence Street, Suite 520, Denver, Colorado 80204-9849.

Hammonds, J.S., Thomas, B.A., Blaylock, B.G., and Thiessen, K.M. 1997. February 1996 Screening Analysis of Radionuclides Released to the Clinch River. Oak Ridge Dose Reconstruction Study. Oak Ridge, Tennessee.

International Atomic Energy Agency (IAEA). 1989. Evaluating the Reliability of Predictions Made Using Environmental Transfer Models. Safety Series No. 100. IAEA, Vienna, Austria.

National Council on Radiation Protection and Measurements (NCRP). 1996. A Guide for Uncertainty Analysis in Dose and Risk Assessments Related to Environmental Contamination. NCRP Commentary No. 14. NCRP. Bethesda, MD.

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5.0 ESTIMATION OF QUANTITIES OF RADIONUCLIDES HISTORICALLY RELEASED FROM WHITE OAK CREEK TO THE CLINCH RIVER

5.1 Introduction

Construction of the X-10 Site began in 1943, and pilot plant operations to produce and chemically separate plutonium started up the following year. Large "Gunite" tanks were built on site to contain all of the liquid radioactive wastes produced from the X-10 operations, which were not expected to continue beyond the end of World War II. The longevity of operations at the X-10 Site, also known as Clinton Laboratory and later Oak Ridge National Laboratory was unanticipated, and expansion of these operations occurred well beyond the end of the war. The capacity of the Gunite tanks was exceeded, and the first waterborne radioactive wastes were released into White Oak Creek in 1944. In anticipation of potential releases of radioactive liquid wastes, White Oak Dam was constructed in 1943 to create a final settling basin for these wastes before releases occurred to the Clinch River. This settling basin is now known as White Oak Lake (Figure 5.1).

From 1944 to 1948, releases of radionuclides to White Oak Creek were primarily from an upstream settling basin constructed near the center of the X-10 Site. In the 1950s, seepage waste pits were constructed to receive X-10's most highly radioactive waste to reduce the quantities being released to the Clinch River. The assumption was that the radionuclides in the liquid waste would be retained by the pit and surrounding soil, while the water would seep into the ground water and emerge as surface water in White Oak Creek. This assumption held reasonably well for ^{137}Cs and other radionuclides with a high affinity for adsorption on soil. However, ^{106}Ru was not retained in the soil, and flowed from the pits with the seepage of ground water. The potential importance of ^{106}Ru was not recognized until 1959, when large quantities of ^{106}Ru were reported in ground water leaving the pits. The highest concentration of ^{106}Ru passing through White Oak Dam was reported in 1960.

In 1955, the gates of White Oak Dam were opened, and the lake was drained. A number of reasons have been postulated for draining the lake, including the belief that the sediments had reached saturation levels with respect to the radionuclides concentrations in the water, reducing the capacity of the sediments to retain additional radionuclides and causing the sediment to become a major secondary source of radionuclides in the water column. In 1956, heavy rains flushed out a portion of the radioactive bottom sediments of the lake bed, resulting in the largest release of ^{137}Cs reported by monitoring instruments at the dam.

Relatively minor radionuclide releases to the White Oak Creek drainage system came from a variety of sources. Starting in 1960, three seepage trenches were built on the X-10 Site to replace the waste disposal pits. The trenches included measures to reduce radionuclide leakage to the ground water. Several test reactors also discharged small quantities of liquid wastes to the White Oak Creek drainage system, and various solid waste burial grounds released some long half-life radionuclides to White Oak Creek. Figure 5.2 presents a time line of historical operations and events relevant to releases of radionuclides from White Oak Creek to the Clinch River.

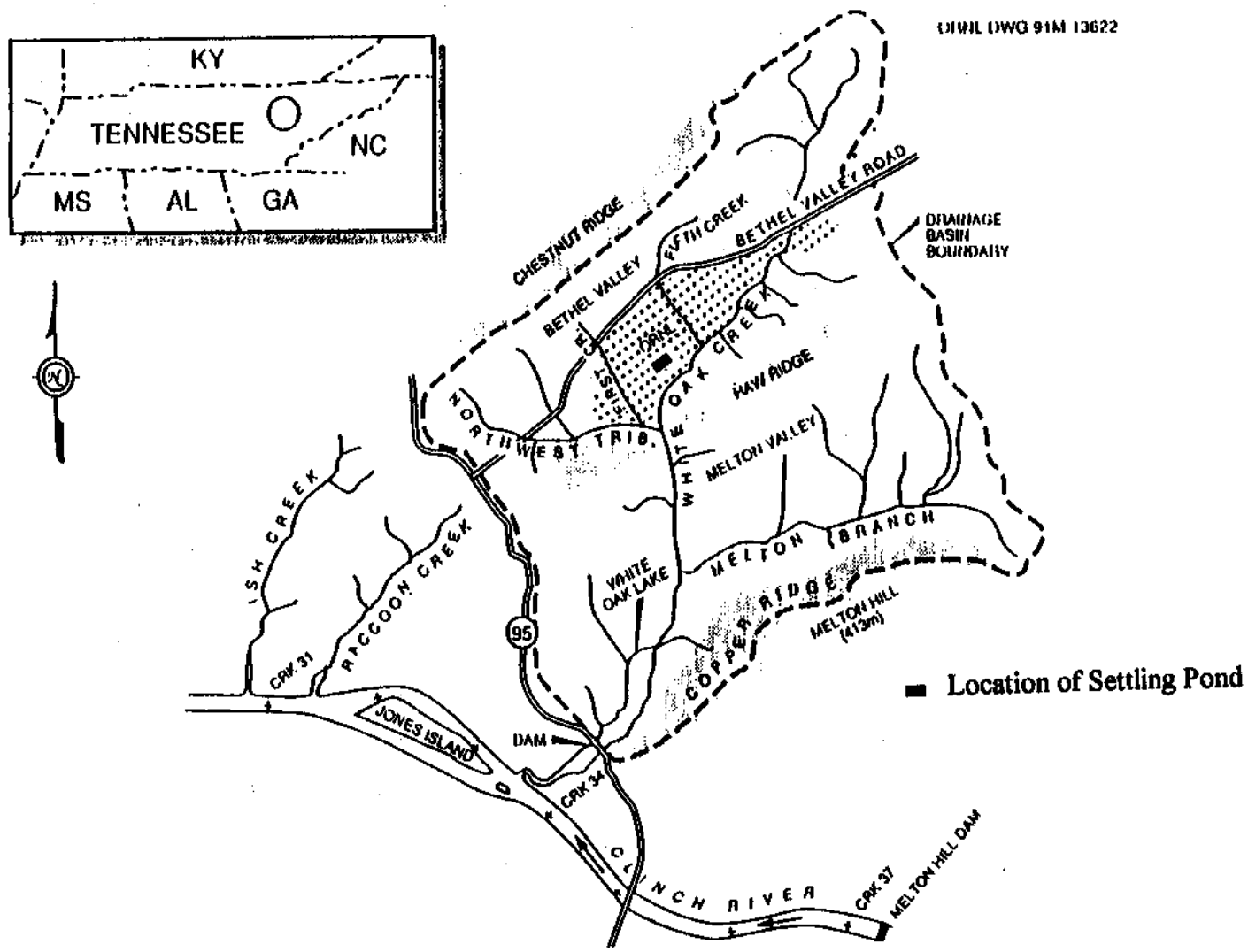


Figure 5.1 Locations of the X-10 site (ORNL), White Oak Creek, White Oak Lake, White Oak Dam, and the Clinch River. White Oak Creek empties into the Clinch River at Mile 20.8, about 2.5 miles below Melton Hill Dam, which was constructed in 1963.

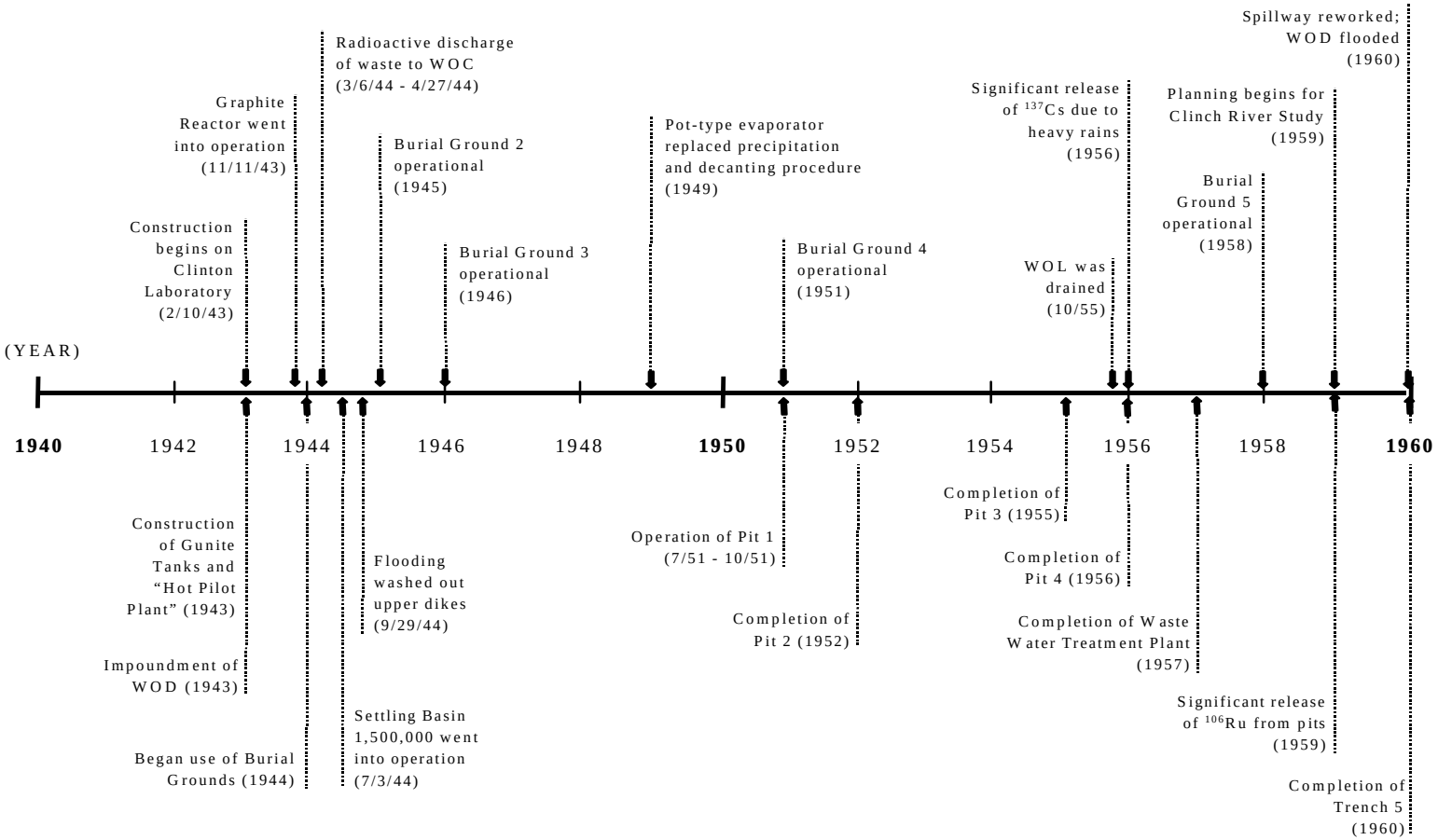


Figure 5.2 Significant events in radioactive waste management and releases from the X-10 site (1940-1960)

Task 4 evaluated radionuclides released from White Oak Creek to the Clinch River, without regard to their sources on the X-10 Site. Because a long historical record of effluent measurements at White Oak Dam was available, a detailed reconstruction of source terms for specific radioactive waste sources on the X-10 Site was not necessary. Use of measurements at White Oak Dam as the basis of this assessment was preferred over reconstruction of releases from individual waste sources for the following reasons:

- The large number of possible individual radionuclide sources on the X-10 Site, including chemical processing of nuclear materials, radioisotope production, reactor operations, and shallow-land burial of radioactive wastes from the Oak Ridge complexes and other sources across the southeast United States;

- The paucity of detailed documentation of the operations listed above and the quantities of radioactive materials that were received, produced, processed, shipped off site, disposed of onsite, or released through X-10 liquid radioactive waste systems; and

- The higher level of uncertainty that would be associated with reconstruction of historical emissions from the individual sources, due to the above considerations, compared to the uncertainties involved with reconstruction of releases based on the available measurements at the point of release from the X-10 Site, White Oak Dam.

This section documents the methods that were used to independently evaluate the quantities of radionuclides that were historically released from White Oak Creek. The approach that was used can be broken into the following steps:

- Information Gathering: This vital initial step included extensive directed searches of records repositories on the Oak Ridge Reservation, interviews of active and retired X-10 workers, and interviews of individuals from other organizations that have been involved with routine or special studies of White Oak Creek or the Clinch River.

- Examination of Past Monitoring and Effluent Reporting Practices: This step involved detailed examination of the methods that were used by the X-10 staff to sample the water that flowed over White Oak Dam and to estimate concentrations of radioactivity or specific radionuclides in that water, associated flow rates, and quantities released. Details of this information gathering process are contained in Section 5.2 and Appendix 5A.

- Identification of Contributors to Uncertainty in Reported Releases: In this step, the practices, errors, or assumptions that were likely to contribute to bias or uncertainty in the officially reported values of radionuclide releases were identified and quantified. This process is described in Section 5.3.

- Application of Correction and Uncertainty Factors to Reported Release Estimates: After thorough review, it was determined in the present investigation that the reported release totals were an appropriate starting point for independent quantification of past releases. A series of correction

factors was then applied to each reported annual release total to account for the effects of each source of bias or uncertainty as described in the previous step. A revised set of emission estimates was produced, along with confidence intervals that reflect the uncertainties associated with the correction and uncertainty factors. This process is described in Section 5.4.

5.2 Historical Information Relevant to Releases from White Oak Creek

The Task 4 information gathering process centered on records located at X-10 Laboratory Records and the X-10 Records Center. Laboratory Records contains more of what would be called "active" records, including copies of thousands of technical reports and Central Files (CF) Memoranda concerning activities at X-10. A large portion of the information relevant to past emissions from White Oak Creek that was located was found in CF memos, log books, and technical reports found in X-10 Laboratory Records. The X-10 Records Center, which contains more of what would be called "retired" or archived records, also yielded some important information such as technical log books.

Following is a summary of information from historical records located by the project team that are relevant to releases from White Oak Creek. This information centers around past sampling and measurement practices.

Within this section, specific documents are often mentioned with their CF Memorandum numbers (such as CF 44-08-346). More information regarding these documents can be found in Appendix 5A, which contains a listing of reports with data relevant to measurement of discharges from White Oak Creek.

Although many reports regarding monitoring of radioactivity in discharges from White Oak Creek were located at X-10 Laboratory Records (see Appendix 5A), a smaller number of report series were found to be most useful for estimating a source term for radionuclide releases. The following documents contained summary level historical data regarding annual releases of radionuclides from the X-10 site:

- Notebook No. 12 by R.A. Lauderdale, January 5, 1949, p.75 (ChemRisk Repository No. 2075)
- Radioactive Waste management at ORNL by F.N. Browder et al., April 14, 1959, p. 54, Table V (ORNL 2601; ChemRisk Repository No. 1990)
- Analysis of Waste Disposal Practice and Control at ORNL by Y. Feige, F.L. Parker and G. Struxness, October 4, 1960, p. 18. Table 2 (ORNL CF-60-8-72; ChemRisk Repository No. 1986)
- Estimate of Radioactivity Release to Clinch River for Period 1944 to 1947, Memorandum from F.L. Parker to D.M. Davis, December 19, 1962 (ChemRisk Repository No. 3434)
- Description of ORNL Liquid Waste Systems, Hazards Evaluation - Volume 3 by F.N. Browder, August 21, 1962, p.13, Table 1 (ORNL-TM-324; ChemRisk Repository No. 680)
- Safety Analysis of Radionuclide Release to the Clinch River by K.E. Cowser and W.S. Snyder, May 1966, p. 6, Table 1 (ORNL-3721; ChemRisk Repository No. 207)

- Waste Management at ORNL by W.D. Burch, September 1972, p. 193, Table 11.9, (ORNL CF-72-9-1; ChemRisk Repository No. 1957)
- Technical Background Information for the Environmental Safety Report, Vol. 4: White Oak Lake and Dam by T.W. Oakes et al., March 1982, p. 128, Table 6.1 (ORNL-5681; ChemRisk Repository No. 156)
- Historical Releases of Radioactivity to the Environment from ORNL by W.F. Ohnesorge, May 1986, pp. 18-19, Table 3 (ORNL/M-135; ChemRisk Repository No. 609)
- Historical Radionuclide Releases from Current DOE Oak Ridge Operations Office Facilities by U.S. Department of Energy, Oak Ridge Operations Office, May 1988, Table 2 (OR-890; ChemRisk Repository No. 446)

In addition, Applied Health Physics reports (1953-84) and Monthly Liquid Waste Disposal reports (1961-1985) were key documents that were located and were used to evaluate the estimates of annual radionuclide releases from White Oak Creek reported in the documents identified above.

Interviews with the following individuals were conducted to identify past monitoring and data reporting practices used by the Area Monitoring and Health Physics groups at ORNL to quantify releases of radionuclides from White Oak Creek.

- H. Abee, ORNL Area Monitoring (ChemRisk Repository No. 2077)
- S. Auerbach, ORNL Environmental Sciences Division (ChemRisk Repository No. 2032)
- C. Barton, ORNL Operations (ChemRisk Repository No. 2024)
- M. Bauer, ORNL Environmental Monitoring (ChemRisk Repository No. 2076)
- W.D. Cottrell, ORNL Area Monitoring (ChemRisk Repository No. 1632, 3020)
- K. Cowser, ORNL Waste Disposal (ChemRisk Repository No. 2023)
- F. Kornegay, ORNL Environmental Monitoring (ChemRisk Repository No. 1520)
- K.Z. Morgan, ORNL Health Physics (ChemRisk Repository No. 3392)
- W.F. Ohnesorge, ORNL Environmental Monitoring (ChemRisk Repository No. 1628)
- J. Phelps, former ORR employee (ChemRisk Repository No. 2226)
- M. Sealand, ORNL Waste Management (ChemRisk Repository No. 2078)

5.2.1 Measurements of Gamma Radiation from Water Samples

Measurements of White Oak Lake effluents were initially conducted with an emphasis on dose rates that people would receive if they were immersed in the contaminated water, rather than an emphasis on quantifying quantities of radioactivity released. This emphasis was apparently based on the initial understanding of the radionuclide composition of the effluents and a belief that external radiation exposures would be more restrictive in complying with the exposure guidelines of the time than internal exposures from ingestion of contaminated water or other exposure pathways. It was thought that this would be the case while the principal radioactivity present were isotopes of columbium (now called niobium) and zirconium rather than barium or strontium (Morgan, 1945). Measurements of gamma radiation from White Oak Dam

water samples became the basis for retrospective estimation by X-10 staff of gross beta releases for 1944 through 1947 (see Section 5.2.2).

The earliest identified water monitoring data for radioactivity at White Oak Dam were collected on July 26, 1944. These data are presented in "Activity of Mud and Water from White Oak Dam" (CF-44-08-376). This one page report tabulates results from single samples collected at White Oak Dam and at seven locations along White Oak Creek. Radioactivity data for both water (in “c/ml/min”) and mud (in “µc/gm”) are presented. Collection or analysis methods are not discussed.

Three subsequent reports presented radioactivity monitoring results for water samples collected at White Oak Dam during the three week period between September 11, 1944 and October 1, 1944 (Clinton Laboratories, 1944a;b;c). These data were collected using the "small container/ large container" immersion monitoring method (Parker, 1944). This monitoring method, specific for gamma emitting radionuclides, yields gamma dose rate measurements (abbreviated “mr/hr” at the time), representing radiation intensities on the basis of simulated immersion in an infinite volume of the contaminated water. The objective of this method was to allow measurement of an indicator of gamma dose rates when it was inconvenient to take measuring equipment to the water supply to actually measure activity in an infinite volume (relative to the ranges of the radiations involved). This method involved immersion of the measuring device (a brass-walled Geiger-Mueller tube) successively at the center of two water-filled containers of radius R and 2R, where R was approximately 3.5 inches. The “dosage-rate at infinite volume” could be calculated from measurements made in the two containers based on the following equation:

$$G = \frac{S^2}{(2S - L) \times 3350}$$

where:

G	=	Dose rate at center of an infinite sphere (mr/hr),
S	=	Count rate in small container (counts/min),
L	=	Count rate in large container (counts/min), and
3350	=	Conversion factor, counts/min to mr/hr

The equation can be rewritten as:

$$G = \frac{2.99 \times 10^{-4} \times S^2}{(2S - L)}$$

According to the methodology described by Parker (CH-1889; 1944), the brass counters used gave 3350 counts/min when exposed to a 1 mr/hr field of radium gamma radiation. According to CH-1889, results prior to July 1944 corresponded well with direct ionization measurements in the X-10 settling basin and White Oak Lake.

A series of reports, for the period October 1944 through March 1948, presents radioactivity monitoring results for water samples collected at the inlet and outlet to the settling basin and at White Oak Dam. In general, samples were collected approximately daily at the inlet and outlet to the settling pond and once per week at White Oak Dam. Individual reports were issued weekly from October 1944 through early 1946 and less frequently (biweekly or monthly) from early 1946 through March 1948. Reports in this series identified in the Central Files database are listed in Appendix 5A.

Samples were collected and measured for gamma activity using the small/large container immersion method. These reports indicate that a thin-walled, one inch brass GM tube was immersed in water samples in small and large containers. The method used to estimate dose rate appears to differ somewhat from that described in CH-1889. Specifically, gamma activity measurements in these reports were reported separately for the small and large containers. Values corresponding to the two containers are presented in counts per minute (c/m) and converted to mr/hr as follows:

$$G_S = (6.2\times10^{-4}) \times \frac{S}{3}$$

$$G_L = (4.1\times10^{-4}) \times \frac{L}{3}$$

here:

G_S	=	dose rate (mr/hr) within the small container,
G_L	=	dose rate (mr/hr) within the large container,
S	=	Count rate in small container (counts/min), and
L	=	Count rate in large container (counts/min).

These values represent radiation intensities on the basis of simulated immersion in an infinite volume, assuming an average gamma ray energy of 0.7 MeV. During early periods (i.e., beginning in 1945), measurements made in the two containers were simply compared and the consistency between the measurements noted, to provide an indication of trends in gamma activity. At later dates (e.g., 1948), the values measured in the two containers were averaged by X-10 workers for reporting.

5.2.2 Measurements of gross beta activity releases

The objectives of the White Oak Dam monitoring program changed between 1944 and 1949. The first attempts to quantify daily releases of radioactivity occurred in October 1947. Between October and December 1947, daily samples of White Oak Lake effluent were analyzed for beta emitting radioactivity without regard for the identity of the radionuclides present, often called gross beta radioactivity (Cottrell, 1948). This monitoring was part of a special study, apparently to test methods for quantifying releases

fromWhite Oak Lake[1]. The methods of this 1947 study were carried into a routine monitoring program for measurement of White Oak Dam releases.

The daily monitoring of radioactive effluent at White Oak Dam consisted of grab sampling the flow over the dam's gate. This sample was then transported to the Health Physics laboratory for quantification of the gross beta radioactivity present through use of an end-window Geiger- Mueller detector system. X-10's monitoring program averaged the gross beta radioactivity concentration on a weekly basis, multiplied this concentration by the amount of flow measured at White Oak Dam for that week, and reported the resulting gross beta curies released for the week. The daily grab sample of White Oak Dam effluent initially consisted of eight gallons, which not only provided the daily gross beta sample but also provided the daily aliquot for monthly composite analysis and provided other X-10 groups with enough volume to conduct waste treatment studies.

Sporadic weekly reports retrieved from X-10 Central Files, starting in 1948, list weekly quantities of radioactivity released from White Oak Dam, in curies. The weekly reports located from 1948 were titled "Waste Monitoring Reports", and progressed into regularly reported weekly "Area Monitoring Reports". These weekly Area Monitoring Reports were recovered for the entire period of 1953-1956. These 1953-1956 documents allowed the reconstruction of historical releases from this period and the verification of the continued use of standard protocols.

Using independent flow rates measured by the United States Geological Survey and the average weekly gross beta concentrations at White Oak Dam, weekly gross beta curie releases were calculated whenever possible. The availability of USGS flow rates and X-10 gross beta concentrations coincided for half of 1953, entire calendar year 1954, and half of 1955. Calculations of weekly gross beta curie releases by the project team agreed well with those reported by the Area Monitoring Group. The release total calculated for calendar year 1954 was within one curie of X-10's reported release for that year (approximately 390 Ci). For 1953 and 1955, some estimation of flow rates was necessary, as USGS data for White Oak Creek could not be located. Weekly variations between the calculations by the project team and X-10 staff likely resulted from the project team using different flow rates for periods for which the values that X-10 staff used could not be located.

Details of the methods and results of the independent reconstruction of releases for 1954 are provided in Appendix 5B.

Retrospective Estimation of Gross Beta Releases for 1944-1947 by X-10 Personnel:

White Oak Lake effluents were not monitored for gross beta radioactivity on a regular basis before late 1947. As a result, curie releases for this time period could not be directly calculated. A 1962 document by Frank L. Parker (1962) of the X-10 Health Physics Division stated:

[1] Personal communication between Brian Caldwell of the project team and W.D. Cottrell, former X-10 health physics/area monitoring worker.

> "Occasionally, the question is raised of the curies of activity released to the Clinch River during the period, 1944 through 1947. Currently, we have need for such values in the Safety Analyses Subcommittee of the Clinch River Study. Unfortunately, direct measurements of stream flow and radionuclide concentration in White Oak Creek are not available to allow calculation of release to the Clinch River during this period. Therefore, estimates of release have been made using the best data available to us."

The 1962 memo described the methods used to estimate gross beta releases for 1944 through 1947 using an observed relationship between annual average gross beta concentrations (μCi/ml) and the annual average gamma dose rate (mr/hr) from 1945 and 1949 through 1960. For each year of the selected time period, the X-10 Health Physics staff calculated the ratio of annual average gross beta concentration to annual average gamma dose rate. The ratios were averaged over the 13-year period to calculate an assumed representative constant that existed between the two measurements. The X-10 Health Physics division multiplied the average ratio from the 13 year period to the annual-average gamma dose rates from 1944 through 1947 to "back-calculate" average annual gross beta concentrations. Table 5.1 presents the data used to develop an average ratio of 1.3×10^{-3} μC/ml per mr/h.

The process used to back-calculate gross beta concentrations can be summarized as follows:

$$\textit{Mean Annual Ratio (for 13 years)} = \frac{\textit{Annual Average Gross Beta Conc.}}{\textit{Annual Average Gamma Dose Rate}}$$

Then, for each year from 1944 through 1947:

$$\textit{Estimated Gross Beta Concentration} = (\textit{Measured Gamma Dose Rate}) \times (\textit{Mean Annual Ratio}$$

Table 5.1 Data Used by X-10 staff to estimate gross beta releases for 1944-1947

Year	Annual Average Gross Beta Concentration (μCi/mL)	Annual Average Gamma Dose Rate (mr/hr)	Annual Ratio, Gross Beta Conc. to Gamma Dose Rate
1945	5.00×10^{-5}	0.027	1.90×10^{-3}
1949	4.00×10^{-5}	0.042	9.60×10^{-4}
1950	1.20×10^{-5}	0.013	9.10×10^{-4}
1951	7.30×10^{-6}	0.027	2.70×10^{-4}
1952	2.50×10^{-5}	0.044	5.70×10^{-4}
1953	2.40×10^{-5}	0.025	9.50×10^{-4}
1954	3.40×10^{-5}	0.029	1.20×10^{-3}
1955	4.40×10^{-5}	0.022	2.00×10^{-3}
1956	4.60×10^{-5}	0.034	1.40×10^{-3}
1957	2.50×10^{-5}	0.019	1.30×10^{-3}
1958	5.50×10^{-5}	0.028	2.00×10^{-3}
1959	9.20×10^{-5}	0.044	2.10×10^{-3}
1960	2.14×10^{-4}	0.165	1.30×10^{-3}

Annual Ratio Summary Statistics

Mean:[1]	1.30×10^{-3} μCi/mL per mr/hr
Standard Deviation:	5.76×10^{-4}
Two Std. Deviations:	1.15×10^{-3} (88% of mean)

[1] Value used by X-10 staff to estimate gross beta releases for 1944-1947 (Parker, 1962).

Appendix 5C presents measured gamma dose rates from 1945, along with a discussion of the completeness of gamma dose rate measurements in general.

5.2.3 Estimation of Radionuclide-Specific Releases

Even after X-10 personnel began reporting releases of specific radionuclides from White Oak Lake, the gross beta radioactivity measurements remained fundamental to the calculation of annual releases of radioactivity to the Clinch River. In 1949, X-10 began apportioning monthly gross beta releases among several specific radionuclides. The monthly composite sample, made up from daily aliquots, was chemically separated to quantify activities associated with selected isotopes. After proper chemical processing and corrections for separation efficiency, the separated volume for each radionuclide was counted to determine the radioactive contribution of each radionuclide. The isotopic contributions were determined by comparing the counts per minute per milliliter of each separated radionuclide to the count rate from the monthly composite sample as a whole. The isotopic contribution of each component was then converted to a percentage and multiplied by the monthly gross beta release, in curies, to determine the monthly curie release of each isotope. These monthly radionuclide-specific curie releases were then summed annually (Ohnesorge, 1986). The isotopes chosen for separation were initially selected based on knowledge of the radionuclides encountered in chemical processing of nuclear materials and later based on radiochemical

analyses and gamma spectroscopy. While documentation of this procedure for separation of gross beta releases to specific radionuclides was located dating back to September of 1944, the procedure did not become an integral part of the monitoring program until 1949.

5.2.4 Measurement of Flow Rates over White Oak Dam

Since basic measurements of radioactivity at White Oak Dam involved quantification of the concentrations of radioactive materials in samples taken from the water flowing over the dam, frequent estimation of the total flow rate over the dam was necessary to convert measured concentrations to release quantities. The methodology used to measure the flow rate over White Oak Dam is described in the report entitled "Measurement Method for White Oak Dam Discharge Volume" (ORNL, 1947). This report describes the apparatus used to measure of discharge volume, the calibration of this apparatus, and the computations of flow rate from the data obtained.

The level of water in the lake was recorded using a clock-driven Bristol recorder, which traced the lake level on a chart; as the lake level increased, the line was traced farther from the center of the chart. The height of the top of the upper gate of the dam, over which the water was discharged, was indicated by a pointer opposite a gauge board marked to show the extent to which the upper gate was open. Zero on the gauge board indicated that the upper gate was entirely closed. Water level indications recorded on the chart were calibrated to estimate an equivalent gauge board reading (2.20 gauge board corresponding to a chart value of 58.0 divisions).

The flow of water through the dam was calculated using the following equation:

$$F = 2/3 \times b \times h^{3/2}\sqrt{2\ g}$$

where:

F	=	Flow ($ft^3\ sec^{-1}$)
b	=	Width of opening (ft) [constant at 4 ft]
h	=	Height of water (ft) [equivalent to the difference between the position of the top of the gate from the gauge board reading and the water level from the chart converted to its equivalent gauge board reading]
g	=	Acceleration of gravity [32 ft sec^{-2}]

Input of known values produces:

$$F = 21.4\ h^{3/2}\ \ \frac{ft^3}{\text{sec}}$$

A somewhat different equation for calculation of flow through the lower gate is presented in "Correction to Report Dated 1/16/46 Entitled: 'Extent of Activity in Mud Washed Through White Oak Dam on January 8, 1946" (ORNL, 1946). According to that report, the above equation is appropriate for estimating flow through the upper gate where the surface of the water is on the same level as the top of the gate opening. However, when the lower gate is considered, a slightly different formula had to be used, as follows:

$$F \text{ ' } \frac{2}{3} \times b \times h \sqrt{2\ gh^{1}}$$

Where:	F	=	Flow ($ft^3\ sec^{-1}$)
	b	=	Width of opening (ft) [4 ft]
	h	=	Height of gate opening (ft) [4 ft]
	g	=	Acceleration of gravity ($ft\ sec^{-2}$) [32 $ft\ sec^{-2}$]
	h^1	=	Distance from water surface to center of gate opening (ft) [8 ft]

Flow through the lower gate according to this equation, added to the flow rate calculated for the upper gate, provided an estimate of the total flow through both gates.

During high flow conditions, the water discharged at White Oak Dam sometimes was higher than the calibrated flow monitoring apparatus. Three reports describing the effects of floods on the discharge of radioactivity over White Oak Dam were identified for the period between 1948 and 1950. The first, "Preliminary Report on Discharges of Radioactivity into White Oak Creek and the Clinch River" (Setter, 1948) presents data collected during the floods of February 12-15, July 14, and November 19 and 28, 1948.

The second report, "Studies on Overflow at White Oak Dam" (Lawler, 1949), presents data collected during the flood of late March 1949. Average curies per day discharged over White Oak Dam and the probable average concentration in the Clinch River are presented for the periods previous to, during, and following the flood, and deviations from the probable average concentration in the Clinch River are calculated. The report indicates that the probable average concentration in the Clinch River was calculated using a dilution factor that represented the ratio of White Oak Dam discharge to the flow of the Clinch River. Average discharge volumes from White Oak Dam are provided for each day of sampling; however, the flow of the Clinch River for corresponding periods is not provided.

The third report “Monitoring of White Oak Discharge Water During Flood of January 30-31, 1950" presents data collected between January 29, 1950 and February 2, 1950, when 5.65 inches of rainfall were recorded at ORNL. During this period, data were collected on an "around the clock schedule" to monitor radioactivity in discharge water from White Oak Dam. Gamma radioactivity was measured from approximately eight gallon samples collected over a 28 hour period. Water level in inches above or below the top of the coffer piling is provided for each interval during which samples were collected. Total curies discharged per day over White Oak Dam (as beta activity) and the probable concentration in the Clinch River, calculated based on the dilution afforded by the Clinch River, are presented for each day that

samples were collected. In addition to beta activity, curies discharged per day as gamma activity are also presented. Radiochemical analyses data for ruthenium, zirconium, niobium, total rare earths, cesium, strontium, and plutonium were tabulated as the percent of total activity.

5.3 Contributors to Uncertainty in Reported Releases

Following detailed review of past X-10 effluent sampling and monitoring practices, a number of factors were identified that could possibly have contributed to the overall uncertainty of X-10's reported releases of radionuclides to various degrees. Based on the review of available White Oak Dam monitoring records and independent reconstruction of release totals where feasible, it was determined that X-10's reported releases should serve as the starting point for evaluation of past release quantities. This decision was also influenced by the fact that only a small fraction of the basic monitoring data (i.e., weekly records of curies released over White Oak Dam) for the period of interest were located by the project team. The majority of these records could not be obtained from X-10 records custodians, and could not be located during directed and systematic document searches of numerous repositories. Without a more complete record of these measurements, independent reconstruction of release quantities from basic data would not be feasible for the majority of the period from 1944 through 1991. For each of the contributors to uncertainty described in the following sections, uncertainty factors were developed and applied to the reported releases to correct for or represent identified errors, biases, and uncertainties as a function of time.

5.3.1 Uncertainty Due to X-10's Retrospective Calculation of Gross Beta Releases (1944-1947)

As described in Section 5.2.2, the X-10 staff estimated gross beta concentrations for 1944 through 1947 using an average of the ratios of annual average gross beta concentrations to the annual average gamma dose rates measured by the large container/small container method. While the X-10 personnel used a constant ratio, the annual ratios were known to vary over the 13 years analyzed (see Table 5.1). The uncertainty from the use of a constant ratio in the back-calculation method was estimated by quantifying the fluctuation of the annual ratios over the 13 years. As shown in Table 5.1, all observations fell between the mean ratio and standard deviation, which suggested that using a normal distribution with the same mean and ±2 times the standard deviation would yield a distribution for the values that would contain all observed ratios in Table 5.1. The increased range would also account for the uncertainty in the extrapolation of the data from 1945 through 1960 to the 1944 through 1947 period.

5.3.2 Uncertainty Due to Scaling of Flow Rates from the Little Chestuee (1944-1947)

After the annual-average gross-beta concentrations for each year from 1944 through 1947 were back-calculated, the X-10 personnel multiplied them by the corresponding annual-average flow rates measured at White Oak Dam. However, flow rates for White Oak Lake at White Oak Dam were also unavailable for this period. To compensate for this, the Health Physics Division contracted with the USGS to estimate White Oak Dam flow rates based on flow rates from a similar surface water system. This process, called “scaling,” is a common method for estimating flow rates from one surface water system to another.

The estimation of flow rates of White Oak Creek at White Oak Dam via scaling of measurements from another surface water system was conducted as follows:

$$Q_{WOC} \text{ ' } Q_G \times \frac{A_{WOC}}{A_G}$$

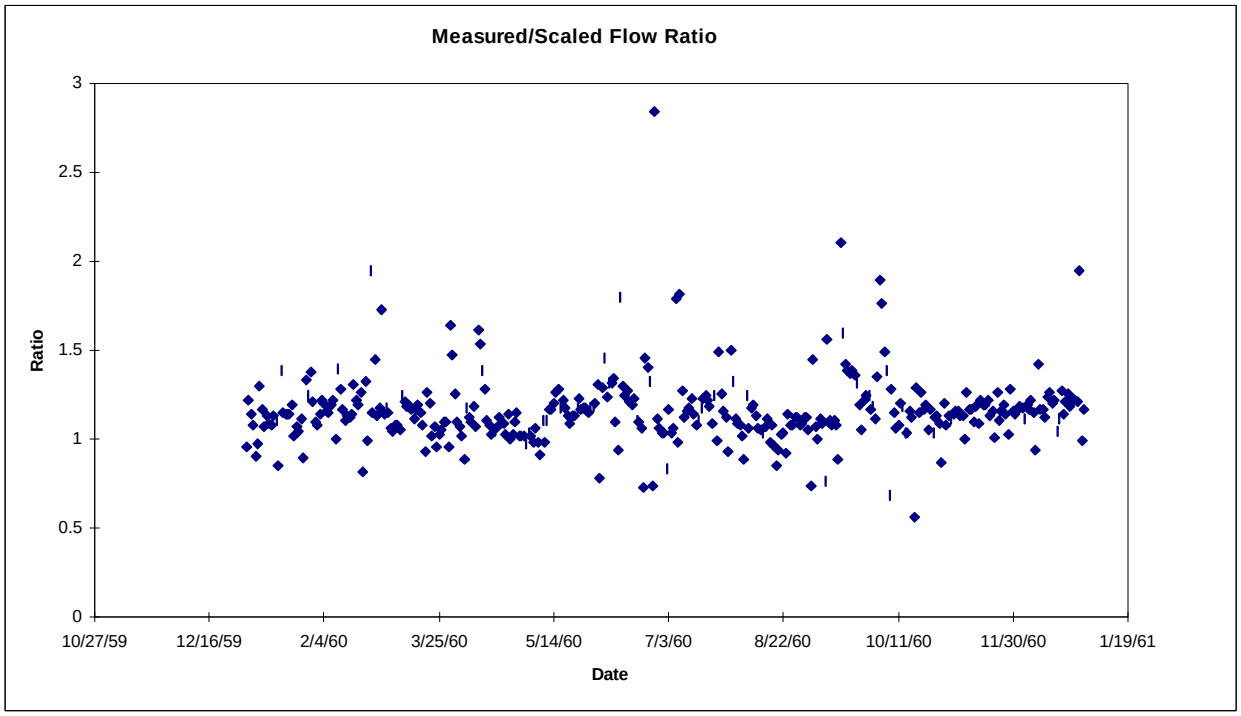
where:

Q_{WOC}	=	Predicted flow rate of White Oak Creek ($ft^3\ s^{-1}$),
Q_G	=	Measured flow rate at gauging station on the other surface water system ($ft^3\ s^{-1}$),
A_{WOC}	=	White Oak Creek catchment area (mi^2), and
A_G	=	Catchment area of the other surface water system (mi^2).

The USGS selected the Little Chestuee River near Cleveland, Tennessee for scaling purposes, because the Little Chestuee had a similar drainage area and geographical setting to that of White Oak Creek and daily flow rates for this system were available for the period in question.[1] However, scaled flow rates from the Little Chestuee River are not exact representations of actual flows discharged at White Oak Lake. Factors that influence differences between actual measured flow rates and scaled flow rates include precipitation differences, White Oak Creek's reception of daily effluents from X-10's settling basin, and the manipulation of White Oak Dam's flow gates for the control of the releases of radioactivity.

In order to estimate the associated uncertainties, scaled flow rates from the Little Chestuee River were compared to actual measured White Oak Lake discharges for periods when both measurements were available (1953 through 1955 and calendar year 1961). The ratios of daily and annual flow were calculated by dividing *measured* White Oak Dam flows by White Oak Dam flow *scaled* from the Little Chestuee River. Values of the daily ratios are plotted in Figure 5.3. The results suggest that flow rates scaled from the Little Chestuee River had a tendency to under- predict the actual flow rates discharged at White Oak Lake (see Figure 5.3). On an annual basis, the scaled flow rates ranged between 72 and 75 percent (most likely 74%) of the measured flow rates. A correction factor was applied to the reported annual releases, ranging between 1.33 to 1.38 and described by a triangular distribution with a most likely value of 1.35.

[1] Personal communication between Brian Caldwell of the project team and Bryan Bradley, Subdistrict Chief of the Water Resources Division, USGS. June 1996.



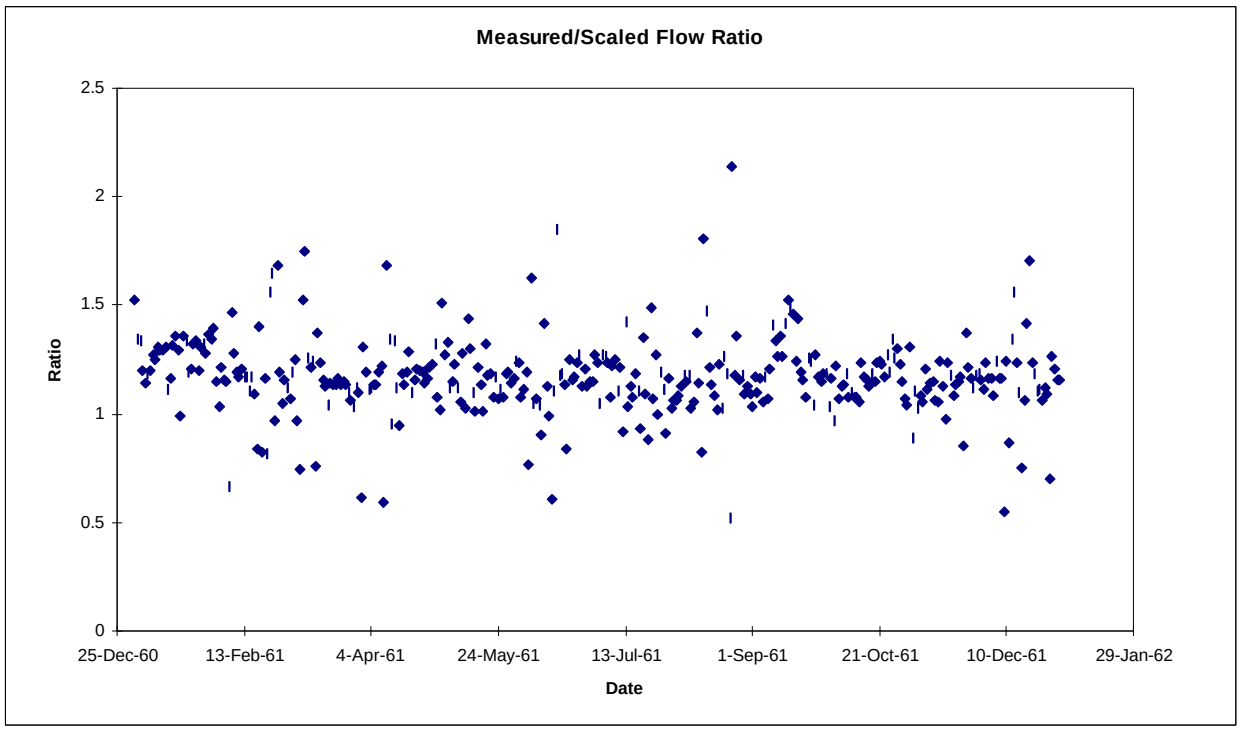


Figure 5.3 Ratios of measured White Oak Dam flow rates to flow rates scaled from the Little Chestuee River.

5.3.3 Uncertainty Due to Non-Proportional Sampling (1944-1960)

As presented earlier in this section, starting in 1947, daily gross beta concentrations were calculated based on analyses of grab samples taken at White Oak Dam. Grab sampling can be a significant contributor to the overall uncertainty of reported releases if contaminant concentrations in the effluent change significantly between sampling periods, or if wastes are released in batches. In contrast, flow-proportional sampling would sample larger volumes during periods of high stream flow to account for the increased total discharge of radionuclides compared to the periods of lower flow at the dam.

Analysis of the daily gross beta concentrations revealed that concentrations released over White Oak Dam, from the 1953-1956 time period, did not fluctuate significantly over short time periods. Even though the White Oak Creek watershed reduced the likelihood of significant fluctuations in concentrations released over the dam, it was felt that an uncertainty factor should be evaluated. The data was not available to compare concentration measurements based on proportional samples with concentrations estimated based on grab sampling. To compensate for this, measured gross beta concentrations and measured flow rates were used to generate *simulated* proportionally sampled concentrations to support evaluation of the potential impacts of grab sampling. The data used for this analysis included the first half of 1953, the second half of 1955, complete calendar years 1954 and 1956, and flooding events in 1948. Weekly grab sampled concentrations were compared to simulated proportional sample concentrations based on the daily measured flow rates and volumes sampled.

The simulated proportional concentrations were divided by the grab-sampled concentrations to calculate monthly ratios. The results of this analysis are shown in Table 5.2. The monthly ratios generated for these periods indicated that grab sampled concentrations may have under or over predicted the true gross beta concentrations. However, the degrees of over or under prediction were small enough that the annual average concentration used to calculate annual gross beta releases would have been almost the same using either technique. However, it was concluded that the variability associated with the monthly concentration should be included as an uncertainty factor to account for the lack of flow proportional sampling. The grab sampling factor was specified as a triangular distribution to reflect the range of ratios observed during the test period. The triangular distribution ranged between 0.76 (the minimum observed ratio) and 1.32 (the maximum observed ratio), with a most likely value of 1.

In 1959, a proportional stream sampler was installed and tested at White Oak Dam. In 1960, the proportional sampler became operational and was used on a regular basis to sample White Oak Dam effluent. For this reason, the use of this uncertainty factor for grab sampling was not required after 1961.

Table 5.2 Comparison of grab sample to simulated proportional sample concentrations.

Period of data collection	Total grab sample conc. ($\mu g\ mL^{-1}$)	Total simulated prop. conc. ($\mu g\ mL^{-1}$)	Annual ratio	Average ratio	Standard deviation	Minimum ratio	Maximum ratio
1948 floods	1.73×10^{-4}	1.68×10^{-4}	0.97	0.97	0.06	0.86	1.01
1954	5.49×10^{-1}	5.84×10^{-1}	1.07	1.05	0.11	0.90	1.33
1953-5	3.73×10^{-1}	3.70×10^{-1}	0.99	0.99	0.11	0.77	1.26
1956	6.64×10^{-1}	6.57×10^{-1}	0.99	1.00	0.05	0.93	1.10

5.3.4 Variability Due to Laboratory Processing of Effluent Samples (1944-1991)

Another contributor to the overall uncertainty of the reported release totals was the laboratory processing used to prepare samples for analysis. The process for determining the radioactivity in each water sample involved several steps. Once the sample was taken at White Oak Dam, it was transported to the Area Monitoring laboratory for several analyses, one of which was gross-beta radioactivity determination. The aliquot used to determine the gross beta concentration was mounted on a watch glass and evaporated to dryness. This watch glass was then placed on a selected shelf of a calibrated radiation detection system, which then quantified the amount of radioactivity present. Variability in the aliquot volume, the evaporation step, and the mounting technique introduced some degree of uncertainty in the final results of the gross beta determination. An X-10 internal analysis of these early standard laboratory processing techniques, conducted in 1945 (Clinton Laboratories, 1946), was utilized to determine the uncertainty with these techniques.

The studies conducted by X-10 personnel compared the analytical results from an analysis of strontium and barium from different technicians, who had never previously conducted the analysis. Based on an evaluation of the studies conducted by X-10 personnel, it was concluded that the referenced study was applicable for obtaining a factor for the uncertainty due to laboratory processing. When the technicians' laboratory results were plotted as histograms, the distributions approximated normal distributions. The details and statistics of the results are shown in Table 5.3.

The relative differences between the upper 97.5^{th} percentile and the mean for strontium and barium were calculated, and the results with the largest percent difference were selected for defining the uncertainty factor. The extraction process for barium had the larger percent difference, at ± 4.7 percent. An increased uncertainty range was not applied, since the experiments involved inexperienced technicians, and the uncertainty would be expected to be small when experienced technicians were used. Also, the maximum percent difference was applied to the annual gross beta estimates, which would then apply to all radionuclides under study.

Table 5.3 Variability of results from laboratory processing.

	Barium Measurement (mg)	**Strontium Measurement (mg)**
	82.1	103
	85.8	107
	86.0	108
	86.5	109
	86.9	109
	86.9	110
	87.1	110
	87.8	110
	88.3	---
	88.7	---
	89.3	---
	90.1	---
Statistics		
Average	87.1	108
Std. Deviation	2.06	2.39
97.5^{th} percentile[1]	92.2	113
% variation[2]	0.0471	0.0463

[1] Estimated assuming normal statistics
[2] Percent variation calculated as 1 - (97.5^{th} percentile/ Average)

Since the laboratory processing techniques utilized by X-10 to determine the gross beta radioactivity present did not change dramatically throughout the history of X-10 operations, this uncertainty factor was applied to each year from 1944 through 1991. The factor was applied to the annual reported gross beta curie releases as a normally distributed function with the ± 4.7 percent differences representative of the 95% subjective confidence interval..

5.3.5 Random Uncertainty Inherent in Counting Radioactive Decays (1944-1991)

Determination of the radioactivity present in an effluent sample analyzed at X-10 also had some degree of uncertainty associated with it, from the variability associated with counting each daily sample. Several factors would have contributed to the overall variability, including the random nature of radioactive decay and the several steps performed during the laboratory processing of samples. As discussed in the previous section, one of the reports documented two separate analyses to determine the variability introduced from using two separate technicians during the laboratory processing. The same document was then used to quantify the uncertainty due to counting variability.

After each technician performed the appropriate sample processing, they quantified the radioactivity present using the same standard calibrated counting equipment. Again, the two sets of counting data were plotted as a histogram, and the results approximated a normal distribution. A component of the uncertainty analysis in radiometric analysis arises from the random uncertainty inherent in counting radioactive decays. In other

words, there is uncertainty involved in estimating the radioactivity of a sample if the counting time is not sufficient to provide a stable estimate of radioactivity. The lower the radioactivity, the longer the time needed for counting. Data from counting of radioactive samples by different technicians from CN-1312 (Clinton Laboratories, 1946) are presented in Table 5.4.

Table 5.4 Variability of results from counting radioactive decays.

	Barium Measurement (counts/min)	**Strontium Measurement (counts/min)**
	1095	3065
	1026	2925
	1053	2800
	1098	2895
	1160	3010
	953	2915
	1115	3350
	1085	2645
	1000	---
	1045	---
	1190	---
Statistics		
Average	1075	2951
Std. Deviation	68.8	206
97.5th percentile[1]	1212	3363
% variation[2]	12.8	14.0

[1] Estimated assuming normal statistics
[2] Percent variation calculated as 1 - (97.5th percentile/ Average)

Assuming that the counting variability was represented by a normal distribution, the mean and standard deviation for both data sets were calculated to determine the individual 95th percent confidence intervals as the region contained within two standard deviations on either side of the mean. The corresponding two standard deviations for barium and strontium represented 13 and 14 percent of their respective means. The larger value of 14 percent, from the strontium data set, was selected as the basis for an uncertainty factor. A normal distribution with lower and upper limits of 0.86 and 1.14, respectively, of a 95% confidence interval, was chosen as representative of the uncertainty. This factor was applied to the gross beta estimate for every year from 1944 through 1991.

5.3.6 Uncertainty with Counting Efficiencies of Measurement Systems (1944-1958)

Another uncertainty identified and quantified was associated with the radiation detection system used by X-10. Specifically, the uncertainty of the efficiency (or the fraction of disintegrations in a radioactive sample that are detected) of the detection system which results in a number of "counts" was questioned. The single channel analyzer system utilized by X-10 personnel was assumed to have an efficiency of 10 percent (i.e.,

the analyzer recorded one "count" for every 10 disintegrations). This efficiency was based on the use of uranyl nitrate hexahydrate (UNH) solution as a calibration standard for the detection system. The UNH calibration solution had an average energy of approximately 400 keV, which was assumed to approximate the average energy of radiations from White Oak Lake effluent. It was verified that the average beta-particle energy of White Oak Lake effluents fluctuated around 400 keV for the first fifteen years of operation, based on an analysis of several isotopic distributions. Therefore, the UNH solution was a reasonable calibration source until 1959, when large amounts of ruthenium-106 leached from waste pits and entered White Oak Lake, changing the average energy of White Oak Lake effluent. However, based on technician logbooks containing general information on the X-10 counting systems, the efficiency of the single channel analyzer used by the Area Monitoring Group fluctuated on a daily basis. Therefore, using an assumed constant efficiency factor of 10 percent for gross-beta determinations would have contributed to the overall uncertainty of reported release quantities.

The variability of the counting efficiency listed in the technician logbooks were used to estimate the uncertainty introduced by using a constant efficiency factor. The information contained in these logbooks showed that actual detector efficiency for the 1944-1958 time period ranged between 9 and 11 percent. Because efficiency factors are used in the denominators of equations to calculate quantities of radioactivity present, the uncertainty factor associated with the 9 to 11 percent actual detector efficiency would range between 0.9 and 1.1. A uniform distribution between 0.9 and 1.1 was used for this factor.

Due to the large amounts of ruthenium-106 leaching from waste pits to White Oak Creek in 1959, the Area Monitoring Group selected different calibration sources and calculated new efficiencies for the counting equipment on a weekly basis to approximate the average energy of the White Oak Lake effluent. The uncertainty due to counter efficiency became relatively insignificant with the selection of different calibration sources and re-calibration of the counting equipment for each determination.

5.3.7 Uncertainties with Flow Measurements from Calibrated Weirs (1948-54 and 1961-91)

As previously mentioned, flow rates at White Oak Dam were unavailable for the 1944-1947 time period. However, flow rates at White Oak Dam after 1947 were regularly monitored by X-10 personnel and/or the USGS based on measurements taken with calibrated weirs. Historical flow rates measured by X-10 personnel were only sporadically located, except for a continuous daily record of flows obtained from 1971-1977. However, periods of continuous records of White Oak Dam flow rates were available through the USGS. One exception to this occurred between 1955-1960, when White Oak Lake was drained.

The accuracy of recorded flow rate measurements introduces another contributor to the overall uncertainty when calculating radioactive releases from White Oak Lake. Fortunately, when the USGS records flow rate information, it also publishes the corresponding estimated percentage error with these measurements. Flow records designated with a qualifier of excellent were treated as within 5 percent of the "true" discharge, those with a qualifier as good were treated as within 10 percent of the "true" discharge, and flow records designated with a fair qualifier were treated as within 15 percent of the "true" discharge at a 95% confidence level (USGS, 1993). Flow records that do not meet the USGS criteria were rated as poor.

When available, these reported ratings were utilized to quantify the uncertainty associated with discharge measurements.

From 1953-1955, flow measurements taken at White Oak Dam were qualified as good, except for high flow periods (150 cfs or above), which were qualified as fair. For the period 1960-1962, flow measurements by the USGS were qualified as good, except for those flows measured between April through July, which were qualified as fair. The uncertainties corresponding to the two qualifiers were combined on an annual basis, based on the contribution of each type of qualifier to the annual volume, to estimate a single uncertainty factor each year.

In order to estimate annual frequencies of high-flow periods (above 150 cfs) and normal flow periods for 1948 through 1952, flow records from the USGS (1953-1955; 1961-1964) were utilized. The total annual water volumes attributed to each measurement quality case, in terms of an annual frequency, were calculated for both high flow periods (over 150 cfs) and normal flow periods for the 1953-1955 and 1960-1964 time periods. The average frequency of high flow periods thus calculated was assumed to represent the 1948-1952 time period. The remaining percentage was used as the normal flow percentage for this time period. These flow condition frequencies were combined annually, weighted by the annual average contribution of each qualifier, to estimate the annual flow uncertainty for 1948 to 1952.

Flow rates required to over-top the pilings of White Oak Dam were estimated using the document written by Kochtitzky and Setter (1950a; b). This document provided rating curves that could be used to determine flow rates that corresponded to White Oak Dam gate heights. When the dam gate was at its highest, the flow required to over-top the pilings was 150 cubic feet per second (cfs). Sufficient data were not available to evaluate the ratings made at White Oak Dam beyond 1963, and therefore ratings of flow from 1964-1991 were assumed to have an uncertainty equal to the magnitude reflected by the rating in 1963 (plus or minus 10 percent uncertainty). However, it should be noted that flow measurements beyond 1983, when significant improvements were made to the flow measuring equipment installed when White Oak Dam was renovated and the spillway locations changed, are expected to be extremely accurate.

5.3.8 Uncertainties with Flow Measurements While White Oak Lake was Drained (1955-1960)

During 1955, White Oak Lake was drained by X-10 personnel, in a process that lasted approximately three months. As a result White Oak Creek flowed through the open dam gates from mid-1955 to mid-1960. While this uninhibited flow could not be measured accurately at White Oak Dam, X-10 personnel estimated the daily flow by combining the measured flow of White Oak Creek (above the dam and above the confluence with Melton Branch) and the flow of Melton Branch from data available from 1960 through 1962. During periods of precipitation, this combined flow was multiplied by 1.16 to compensate for the drainage area between Melton Branch and White Oak Dam (Struxness, 1960). To estimate the uncertainty with these flow rate estimates, two separate series of comparisons were made using the available 1960-1962 daily USGS data. First, the simple combinations of White Oak Creek and Melton Branch discharges were compared to the measured White Oak Dam discharges on a daily basis. This comparison illustrated the uncertainty in predicting White Oak Dam flows without compensating for precipitation differences. Measured daily White Oak Dam discharges were then compared with the daily combination flow estimates

corrected for precipitation differences using the factor of 1.16. This second set of comparisons illustrated the uncertainty in predicting White Oak Dam flows with precipitation differences considered.

It was assumed that all flows that averaged over 15 cfs occurred due to precipitation events. This 15 cfs was a bench mark based on the annual average flow rate measured at White Oak Dam, and therefore any flows greater than this annual average were assumed to have been caused by precipitation. For every daily flow measured at White Oak Dam that was above 15 cfs during the period that the lake was drained, the combination flow (from White Oak Creek and Melton Branch) for that day was multiplied by 1.16 before calculating the daily ratio. Each estimated daily flow rate was compared to the measured daily White Oak Dam discharge, and the total annual volume of each flow estimate was compared with the measured annual total volume.

The annual averages of daily flow ratios of measured to estimated flow rates from the comparisons including correction for precipitation, consistently averaged 1.07, and for the uncorrected combination flow ratios they consistently averaged 1.16. Thus, the uncorrected flow rates are associated with upper-bound estimates of uncertainty, having consistently under predicted White Oak Dam flows by 16 percent. The actual value of the ratio is expected to be somewhere between 1.07 and 1.16. Since direct measurement data were not not available from the period of interest, it is possible that X-10's estimated flow rates may have predicted White Oak Dam's flow closer to a ratio of one. Therefore, an uncertainty factor to account for the uncertainty introduced in the reported releases was applied by using a triangular distribution between 1.00 and 1.16, with a most likely value of 1.07.

5.4 Estimation of Quantities of Radionuclides Historically Released

Eight uncertainty or correction factors were specified by the project team for application to the reported annual release totals**.** Their identity and periods of application are depicted in Table 5.5.

The annual curie releases reported by the X-10 personnel were reproduced, thus validating the protocols and methods utilized by X-10 personnel to calculate annual releases. The sources of uncertainties were also identified and quantified with the protocols and methods presented in Section 5.3 in an effort to generate a range of possible releases for each year through uncertainty analysis. The next process was to apply the corresponding factors, as estimated above, to the reported gross beta curie releases in order to generate a range of possible releases for each year. The uncertainties associated with the individual uncertainty or correction factors were propagated to the source term using Monte Carlo simulation with Latin Hypercube Sampling. The Task 4 source term calculations generated 1000 alternate annual gross beta curie releases, using the equation described below:

$$\text{Result} = (\text{Reported Value}) \times (CF_1) \times (CF_2) \ldots \text{ etc.}$$

Where CF_n represents one of the successive uncertainty or correction factors described above and listed in Table 5.5.

Table 5.5 Summary of applicability of uncertainty or correction factors

Source of Uncertainty or Bias	Years of Applicability of Associated Factor				
	1944-47	1948-54	1955-58	1959-60	1961-91
Back-fitting of gross beta releases based on gamma measurements	X				
Scaling of early Creek flows from the Little Chestuee	X				
Creek flow rate measurements		X			X
Flow estimation when Lake drained			X	X	
Non-proportional sampling	X	X	X	X	
Laboratory processing	X	X	X	X	X
Detector efficiency	X	X	X		
Counting statistics	X	X	X	X	X

Following each simulation, the 2.5th and 97.5th percentile values were used as the lower and upper confidence bounds on the gross beta curies released, respectively. The median (50th percentile) value of each iteration was selected as the central estimate of gross beta curies released.

After all forty eight years of releases were estimated using the appropriate uncertainty and correction factors, the lower bound, central value, and upper bound curie release values were distributed among eleven isotopes. A discussion of this distribution is provided in the next section.

Apportionment of Radioactivity to Specific Radionuclides

The procedures outlined in Section 5.2.3 by which X-10 personnel distributed radioactivity among several isotopes was common practice before the age of environmental gamma spectroscopy. Isotopic analyses were conducted monthly and considered separation efficiency, energy differences, decay correction (when necessary), self absorption, back scattering, and detector efficiency. The chemical separations were also improved, as more research and development of these processes became available from X-10. From the documentation collected, sporadic monthly isotopic analyses from 1944-1947, and from 1953-1956 and 1959-1960, it was concluded that the procedures were valid and the reported isotopic percentages were the best available. Therefore, the isotopic-specific fractions reported (for individual isotopes or radionuclides), starting in 1949, were used to distribute the estimated gross beta radioactivity among individual radionuclides for the years 1949 to 1991.

However, due to lack of data in isotope-specific fraction for the 1944-1948 period, another method had to be used to apportion the annual gross beta estimates for these years among individual radionuclides. Even though isotopic distributions were not conducted on a regular basis before 1949, several reports of sporadic monthly composite analyses for the 1944-1948 period were obtained. These reports were incomplete, sometimes containing only four isotopes and other times containing up to 13 isotopes, but they provided valuable data from these early years. It was decided to develop a method that would include the historical relevance of early distributions and would also account for the numerous uncertainties with the incomplete record for this time period. A range of values representing the fraction of each isotope contained in the releases was developed in order to distribute the annual gross beta releases. Available distributions between 1944-1947 and the annual isotopic distributions reported from 1949-1952 were used to develop these ranges of values. Annual isotopic fractions from 1949-1952 were included in this analysis due to the proximity in time (to the 1944-1949 period) and due to the similarity of the procedures and methods employed during the two periods. The annual isotopic fractions from this period were also compared to the individual isotopic fractions reported between 1944 and 1947.

The range for any radionuclide was defined from the minimum and maximum values of the isotopic fraction of that radionuclide reported during the 1944-1952 time period. A uniform distribution was selected for each radionuclide using the minimum and maximum fractions reported during the 1944-1952 period. Table 5.6 presents the minimum and maximum values of the isotopic fraction of each radionuclide used for the 1944-1949 period. Annual estimates of the amount of each radionuclide released was then obtained by multiplying the gross beta releases by the uniform distributions assigned to the fraction of that radionuclide.

Table 5.6 Range of isotopic fractions used to apportion the gross beta releases among individual radionuclides during the 1944-49 period using a uniform distribution.

Radionuclide	Minimum Fraction of Gross Beta	Maximum Fraction of Gross Beta
^{140}Ba	0.01	0.17
^{95}Nb	0.01	0.54
^{90}Sr	0.15	0.34
^{95}Zr	0.15	0.40
^{137}Cs	0.05	0.52
^{131}I	0.01	0.18
^{106}Ru	0.07	0.18
TRE[1]	0.11	0.16
^{144}Ce	0.01	0.11
^{137}Cs	0.006	0.082

[1] Total Rare Earths

Details regarding the methods and data used by the project team to apportion gross beta releases from 1944 through 1948 to specific radionuclides based on isotopic measurements from 1944 through 1952 are presented in Appendix 5D. That appendix also presents the results of an evaluation performed to confirm the annual isotopic percentages reported by X-10 personnel after 1948.

5.5 Results and Sensitivity Analysis

Results of the Task 4 evaluation of the quantities of radionuclides historically released from White Oak Creek to the Clinch River are presented in Table 5.7. A sensitivity analysis was performed to determine, for each year, the factors that contribute significantly to the uncertainty in the estimate of the annual release rate. Table 5.8 presents the three most significant contributors to uncertainty for 1944-1947. In general, the dominant contributors to uncertainty in the source term were (1) back-fitting of gross beta releases based on gamma measurements prior to 1947, (2) nonproportional, grab sampling conducted between 1944 and 1960, and (3) flow rate measurements.

Table 5.7 Independently Estimated Releases from White Oak Creek (Ci)[1]

	Gross Beta			Cs-137			Ru-106		
Year	2.5 Percentile	Central Value	97.5 Percentile	2.5 Percentile	Central Value	97.5 Percentile	2.5 Percentile	Central Value	97.5 Percentile
1944	3.2E+02	8.4E+02	1.4E+03	2.9E+01	1.4E+02	3.3E+02	2.3E+01	6.5E+01	1.3E+02
1945	2.8E+02	6.9E+02	1.2E+03	2.6E+01	1.1E+02	2.8E+02	1.8E+01	5.4E+01	1.1E+02
1946	5.2E+02	1.3E+03	2.2E+03	5.0E+01	2.0E+02	5.1E+02	3.5E+01	9.5E+01	2.1E+02
1947	1.2E+02	2.8E+02	4.7E+02	8.5E+00	4.4E+01	1.1E+02	7.3E+00	2.1E+01	4.7E+01
1948	3.7E+02	5.1E+02	7.0E+02	2.2E+01	8.8E+01	1.7E+02	2.0E+01	3.9E+01	7.0E+01
1949	5.3E+02	7.4E+02	1.0E+03	5.7E+01	8.0E+01	1.1E+02	8.1E+01	1.1E+02	1.6E+02
1950	1.4E+02	2.0E+02	2.7E+02	1.4E+01	2.0E+01	2.7E+01	1.7E+01	2.4E+01	3.2E+01
1951	7.5E+01	1.0E+02	1.4E+02	1.5E+01	2.1E+01	2.8E+01	1.3E+01	1.9E+01	2.6E+01
1952	1.6E+02	2.2E+02	3.0E+02	7.3E+00	1.0E+01	1.4E+01	1.1E+01	1.6E+01	2.1E+01
1953	2.4E+02	3.1E+02	4.1E+02	5.0E+00	6.6E+00	8.7E+00	2.0E+01	2.7E+01	3.5E+01
1954	3.0E+02	4.0E+02	5.2E+02	1.7E+01	2.3E+01	3.0E+01	8.7E+00	1.1E+01	1.5E+01
1955	3.6E+02	4.8E+02	6.4E+02	5.2E+01	7.0E+01	9.2E+01	2.6E+01	3.4E+01	4.5E+01
1956	4.8E+02	6.4E+02	8.6E+02	1.4E+02	1.9E+02	2.5E+02	2.4E+01	3.2E+01	4.3E+01
1957	3.3E+02	4.4E+02	5.8E+02	7.4E+01	9.9E+01	1.3E+02	5.0E+01	6.7E+01	8.8E+01
1958	4.5E+02	6.0E+02	8.0E+02	4.6E+01	6.1E+01	8.1E+01	3.5E+01	4.7E+01	6.2E+01
1959	7.8E+02	1.0E+03	1.3E+03	6.3E+01	8.4E+01	1.1E+02	4.3E+02	5.7E+02	7.5E+02
1960	1.8E+03	2.4E+03	3.1E+03	2.6E+01	3.4E+01	4.4E+01	1.6E+03	2.1E+03	2.7E+03
1961	1.9E+03	2.2E+03	2.6E+03	1.3E+01	1.5E+01	1.7E+01	1.7E+03	2.0E+03	2.3E+03
1962	1.2E+03	1.4E+03	1.7E+03	4.8E+00	5.6E+00	6.5E+00	1.2E+03	1.4E+03	1.6E+03
1963	4.0E+02	4.7E+02	5.4E+02	3.1E+00	3.6E+00	4.2E+00	3.7E+02	4.3E+02	5.0E+02
1964	2.0E+02	2.3E+02	2.7E+02	5.2E+00	6.1E+00	7.0E+00	1.6E+02	1.9E+02	2.2E+02
1965	8.1E+01	9.5E+01	1.1E+02	1.8E+00	2.1E+00	2.4E+00	5.9E+01	6.9E+01	8.0E+01
1966	4.1E+01	4.8E+01	5.5E+01	1.4E+00	1.6E+00	1.8E+00	2.5E+01	2.9E+01	3.3E+01
1967	3.4E+01	4.0E+01	4.6E+01	2.3E+00	2.7E+00	3.1E+00	1.5E+01	1.7E+01	2.0E+01
1968	1.4E+01	1.6E+01	1.9E+01	9.5E-01	1.1E+00	1.3E+00	4.3E+00	5.0E+00	5.8E+00
1969	1.1E+01	1.3E+01	1.5E+01	8.5E-01	9.9E-01	1.1E+00	1.7E+00	2.0E+00	2.3E+00
1970	1.2E+01	1.4E+01	1.6E+01	1.8E+00	2.0E+00	2.4E+00	8.8E-01	1.0E+00	1.2E+00
1971	7.7E+00	9.0E+00	1.0E+01	8.5E-01	9.9E-01	1.1E+00	4.2E-01	5.0E-01	5.7E-01
1972	1.3E+01	1.5E+01	1.7E+01	1.7E+00	2.0E+00	2.3E+00	4.4E-01	5.1E-01	5.9E-01
1973	9.6E+00	1.1E+01	1.3E+01	1.7E+00	2.0E+00	2.3E+00	6.0E-01	7.0E-01	8.1E-01
1974	6.8E+00	8.0E+00	9.2E+00	8.6E-01	9.9E-01	1.1E+00	1.7E-01	2.0E-01	2.3E-01
1975	7.7E+00	9.0E+00	1.0E+01	5.3E-01	6.2E-01	7.2E-01	2.7E-01	3.2E-01	3.6E-01
1976	5.5E+00	6.4E+00	7.4E+00	1.8E-01	2.0E-01	2.4E-01	1.8E-01	2.0E-01	2.4E-01
1977	3.2E+00	3.8E+00	4.4E+00	1.7E-01	2.0E-01	2.3E-01	1.7E-01	2.0E-01	2.3E-01
1978	2.3E+00	2.7E+00	3.1E+00	2.3E-01	2.8E-01	3.2E-01	1.6E-01	1.8E-01	2.1E-01
1979	2.6E+00	3.0E+00	3.5E+00	1.7E-01	1.9E-01	2.2E-01	8.3E-02	9.6E-02	1.1E-01
1980	2.1E+00	2.5E+00	2.9E+00	5.0E-01	5.9E-01	6.8E-01	0.0E+00	0.0E+00	0.0E+00
1981	2.1E+00	2.5E+00	2.9E+00	1.7E-01	2.0E-01	2.3E-01	8.2E-02	9.8E-02	1.1E-01
1982	4.7E+00	5.5E+00	6.3E+00	1.3E+00	1.5E+00	1.8E+00	1.7E-01	2.0E-01	2.4E-01
1983	3.2E+00	3.8E+00	4.4E+00	1.0E+00	1.2E+00	1.4E+00	1.7E-01	2.0E-01	2.3E-01
1984	3.2E+00	3.7E+00	4.3E+00	5.2E-01	6.1E-01	7.1E-01	1.8E-01	2.0E-01	2.4E-01
1985	3.4E+00	4.0E+00	4.6E+00	3.4E-01	4.0E-01	4.6E-01	5.8E-03	6.8E-03	7.8E-03
1986	2.9E+00	3.4E+00	3.9E+00	8.7E-01	1.0E+00	1.2E+00	0.0E+00	0.0E+00	0.0E+00
1987	1.7E+00	2.0E+00	2.3E+00	5.3E-01	6.2E-01	7.1E-01	0.0E+00	0.0E+00	0.0E+00
1988	1.4E+00	1.6E+00	1.8E+00	3.5E-01	4.0E-01	4.8E-01	0.0E+00	0.0E+00	0.0E+00
1989	3.6E+00	4.2E+00	4.8E+00	1.0E+00	1.2E+00	1.4E+00	0.0E+00	0.0E+00	0.0E+00
1990	3.7E+00	4.3E+00	5.0E+00	9.3E-01	1.1E+00	1.3E+00	0.0E+00	0.0E+00	0.0E+00
1991	3.8E+00	4.5E+00	5.2E+00	1.5E+00	1.7E+00	2.0E+00	0.0E+00	0.0E+00	0.0E+00

Table 5.7 Independently Estimated Releases from White Oak Creek (Ci)[1]

	Sr-90			Co-60			Ce-144		
Year	2.5 Percentile	Central Value	97.5 Percentile	2.5 Percentile	Central Value	97.5 Percentile	2.5 Percentile	Central Value	97.5 Percentile
1944	4.4E+01	1.2E+02	2.6E+02	0.0E+00	0.0E+00	0.0E+00	5.5E+00	2.9E+01	7.8E+01
1945	3.8E+01	1.0E+02	2.3E+02	0.0E+00	0.0E+00	0.0E+00	5.1E+00	2.4E+01	6.4E+01
1946	6.8E+01	1.9E+02	3.9E+02	0.0E+00	0.0E+00	0.0E+00	8.5E+00	4.9E+01	1.2E+02
1947	1.6E+01	4.1E+01	8.3E+01	1.1E+00	6.8E+00	2.0E+01	1.8E+00	9.4E+00	2.7E+01
1948	4.2E+01	7.6E+01	1.3E+02	2.5E+00	1.3E+01	3.0E+01	3.8E+00	1.9E+01	4.1E+01
1949	1.1E+02	1.6E+02	2.1E+02	5.5E+00	3.2E+01	6.7E+01	1.3E+01	1.9E+01	2.5E+01
1950	2.8E+01	4.0E+01	5.4E+01	1.5E+00	8.5E+00	1.8E+01	3.5E+00	4.9E+00	6.8E+00
1951	2.1E+01	3.0E+01	4.1E+01	8.0E-01	4.6E+00	9.4E+00	1.9E+00	2.2E+00	2.6E+00
1952	5.3E+01	7.4E+01	1.0E+02	1.7E+00	9.6E+00	2.0E+01	1.7E+01	2.4E+01	3.2E+01
1953	1.0E+02	1.3E+02	1.8E+02	2.5E+00	1.4E+01	2.8E+01	5.2E+00	6.9E+00	9.1E+00
1954	1.1E+02	1.4E+02	1.9E+02	3.1E+00	1.7E+01	3.6E+01	1.9E+01	2.5E+01	3.3E+01
1955	7.7E+01	1.0E+02	1.4E+02	5.4E+00	7.3E+00	9.6E+00	7.0E+01	9.4E+01	1.2E+02
1956	8.3E+01	1.1E+02	1.5E+02	3.8E+01	5.1E+01	6.8E+01	4.8E+01	6.5E+01	8.7E+01
1957	6.9E+01	9.2E+01	1.2E+02	4.0E+00	5.3E+00	7.0E+00	1.1E+01	1.5E+01	1.9E+01
1958	1.2E+02	1.7E+02	2.2E+02	7.2E+00	9.7E+00	1.3E+01	2.5E+01	3.3E+01	4.4E+01
1959	5.0E+01	6.6E+01	8.6E+01	6.4E+01	8.5E+01	1.1E+02	4.0E+01	5.3E+01	6.9E+01
1960	2.4E+01	3.1E+01	4.1E+01	6.0E+01	8.0E+01	1.0E+02	2.2E+01	2.9E+01	3.8E+01
1961	1.9E+01	2.2E+01	2.6E+01	2.7E+01	3.1E+01	3.6E+01	3.8E+00	4.5E+00	5.2E+00
1962	8.0E+00	9.3E+00	1.1E+01	1.2E+01	1.4E+01	1.6E+01	9.9E-01	1.2E+00	1.3E+00
1963	6.4E+00	7.5E+00	8.7E+00	1.2E+01	1.4E+01	1.6E+01	1.3E+00	1.5E+00	1.7E+00
1964	5.6E+00	6.5E+00	7.6E+00	1.3E+01	1.5E+01	1.7E+01	2.6E-01	3.0E-01	3.5E-01
1965	2.9E+00	3.4E+00	3.9E+00	1.0E+01	1.2E+01	1.4E+01	9.0E-02	1.0E-01	1.2E-01
1966	2.6E+00	3.0E+00	3.5E+00	6.0E+00	7.0E+00	8.1E+00	8.6E-02	1.0E-01	1.2E-01
1967	4.4E+00	5.1E+00	5.9E+00	2.6E+00	3.0E+00	3.5E+00	1.7E-01	2.0E-01	2.3E-01
1968	2.4E+00	2.8E+00	3.2E+00	8.6E-01	1.0E+00	1.2E+00	2.6E-02	3.0E-02	3.5E-02
1969	2.5E+00	2.9E+00	3.4E+00	8.4E-01	9.9E-01	1.1E+00	1.7E-02	1.9E-02	2.2E-02
1970	3.5E+00	4.1E+00	4.7E+00	8.7E-01	1.0E+00	1.2E+00	5.2E-02	6.2E-02	7.1E-02
1971	2.5E+00	3.0E+00	3.4E+00	8.4E-01	9.9E-01	1.1E+00	4.2E-02	4.9E-02	5.7E-02
1972	5.2E+00	6.0E+00	7.0E+00	8.6E-01	1.0E+00	1.2E+00	2.6E-02	3.0E-02	3.5E-02
1973	5.9E+00	7.0E+00	8.0E+00	8.5E-01	9.9E-01	1.1E+00	1.7E-02	2.0E-02	2.3E-02
1974	5.1E+00	5.9E+00	6.9E+00	5.0E-01	5.9E-01	6.8E-01	1.7E-02	2.0E-02	2.3E-02
1975	6.2E+00	7.2E+00	8.3E+00	4.4E-01	5.1E-01	5.9E-01	0.0E+00	0.0E+00	0.0E+00
1976	4.3E+00	5.0E+00	5.8E+00	7.7E-01	9.1E-01	1.0E+00	0.0E+00	0.0E+00	0.0E+00
1977	2.5E+00	3.0E+00	3.4E+00	3.4E-01	4.0E-01	4.6E-01	0.0E+00	0.0E+00	0.0E+00
1978	1.6E+00	1.8E+00	2.1E+00	3.1E-01	3.7E-01	4.2E-01	0.0E+00	0.0E+00	0.0E+00
1979	2.0E+00	2.3E+00	2.6E+00	3.2E-01	3.8E-01	4.4E-01	0.0E+00	0.0E+00	0.0E+00
1980	1.3E+00	1.5E+00	1.7E+00	3.3E-01	3.9E-01	4.5E-01	0.0E+00	0.0E+00	0.0E+00
1981	1.3E+00	1.5E+00	1.7E+00	5.9E-01	6.9E-01	7.9E-01	0.0E+00	0.0E+00	0.0E+00
1982	2.3E+00	2.7E+00	3.1E+00	8.6E-01	1.0E+00	1.2E+00	0.0E+00	0.0E+00	0.0E+00
1983	1.8E+00	2.1E+00	2.4E+00	2.6E-01	3.0E-01	3.5E-01	0.0E+00	0.0E+00	0.0E+00
1984	2.2E+00	2.6E+00	3.0E+00	1.7E-01	2.0E-01	2.3E-01	0.0E+00	0.0E+00	0.0E+00
1985	2.6E+00	3.0E+00	3.5E+00	5.1E-01	6.0E-01	6.9E-01	0.0E+00	0.0E+00	0.0E+00
1986	1.6E+00	1.8E+00	2.1E+00	4.6E-01	5.4E-01	6.3E-01	0.0E+00	0.0E+00	0.0E+00
1987	1.1E+00	1.2E+00	1.4E+00	1.1E-01	1.3E-01	1.5E-01	0.0E+00	0.0E+00	0.0E+00
1988	9.5E-01	1.1E+00	1.3E+00	6.1E-02	7.2E-02	8.3E-02	0.0E+00	0.0E+00	0.0E+00
1989	2.4E+00	2.9E+00	3.3E+00	1.1E-01	1.3E-01	1.5E-01	0.0E+00	0.0E+00	0.0E+00
1990	2.6E+00	3.1E+00	3.6E+00	9.9E-02	1.2E-01	1.3E-01	0.0E+00	0.0E+00	0.0E+00
1991	2.3E+00	2.7E+00	3.1E+00	1.0E-01	1.2E-01	1.3E-01	0.0E+00	0.0E+00	0.0E+00

Table 5.7 Independently Estimated Releases from White Oak Creek (Ci)[1]

	Zr-95			Nb-95			I-131		
Year	2.5 Percentile	Central Value	97.5 Percentile	2.5 Percentile	Central Value	97.5 Percentile	2.5 Percentile	Central Value	97.5 Percentile
1944	5.0E+01	1.4E+02	3.1E+02	1.1E+01	1.3E+02	3.3E+02	6.5E+00	4.5E+01	1.3E+02
1945	4.2E+01	1.2E+02	2.4E+02	1.1E+01	1.1E+02	3.0E+02	5.2E+00	3.8E+01	1.1E+02
1946	7.2E+01	2.1E+02	4.4E+02	1.7E+01	2.0E+02	5.2E+02	1.0E+01	6.8E+01	1.9E+02
1947	1.7E+01	4.5E+01	9.3E+01	4.0E+00	4.2E+01	1.1E+02	2.1E+00	1.5E+01	4.1E+01
1948	4.4E+01	8.4E+01	1.5E+02	8.0E+00	8.5E+01	1.7E+02	4.8E+00	2.9E+01	6.6E+01
1949	1.3E+02	1.9E+02	2.5E+02	1.6E+01	2.3E+01	3.2E+01	5.7E+01	8.0E+01	1.1E+02
1950	1.1E+01	1.5E+01	2.1E+01	3.1E+01	4.3E+01	6.0E+01	1.4E+01	2.0E+01	2.7E+01
1951	3.4E+00	4.7E+00	6.4E+00	1.6E+00	2.2E+00	3.0E+00	1.3E+01	1.9E+01	2.6E+01
1952	1.4E+01	2.0E+01	2.7E+01	1.3E+01	1.9E+01	2.6E+01	1.5E+01	2.1E+01	2.8E+01
1953	5.9E+00	7.8E+00	1.0E+01	2.8E+00	3.8E+00	5.0E+00	1.7E+00	2.2E+00	2.9E+00
1954	1.1E+01	1.4E+01	1.9E+01	7.2E+00	9.5E+00	1.2E+01	2.7E+00	3.6E+00	4.7E+00
1955	4.3E+00	5.8E+00	7.7E+00	4.7E+00	6.3E+00	8.3E+00	5.8E+00	7.8E+00	1.0E+01
1956	1.0E+01	1.4E+01	1.8E+01	1.2E+01	1.7E+01	2.2E+01	2.9E+00	3.9E+00	5.2E+00
1957	1.9E+01	2.6E+01	3.4E+01	5.9E+00	7.9E+00	1.1E+01	9.9E-01	1.3E+00	1.8E+00
1958	5.0E+00	6.7E+00	8.8E+00	5.0E+00	6.7E+00	8.8E+00	6.8E+00	9.1E+00	1.2E+01
1959	2.3E+01	3.0E+01	3.9E+01	2.5E+01	3.3E+01	4.3E+01	3.9E-01	5.2E-01	6.7E-01
1960	3.1E+01	4.1E+01	5.3E+01	3.8E+01	5.1E+01	6.6E+01	4.4E+00	5.8E+00	7.6E+00
1961	1.7E+01	2.0E+01	2.3E+01	5.9E+01	6.9E+01	8.0E+01	3.1E+00	3.6E+00	4.1E+00
1962	1.8E+00	2.2E+00	2.5E+00	6.5E+00	7.6E+00	8.8E+00	3.1E-01	3.6E-01	4.2E-01
1963	2.8E-01	3.3E-01	3.8E-01	6.0E-01	7.0E-01	8.1E-01	3.6E-01	4.2E-01	4.9E-01
1964	1.4E-01	1.6E-01	1.9E-01	6.0E-02	7.0E-02	8.1E-02	2.4E-01	2.8E-01	3.2E-01
1965	2.8E-01	3.3E-01	3.8E-01	2.8E-01	3.3E-01	3.8E-01	1.7E-01	2.0E-01	2.3E-01
1966	5.8E-01	6.7E-01	7.8E-01	5.8E-01	6.7E-01	7.8E-01	2.1E-01	2.4E-01	2.8E-01
1967	4.1E-01	4.8E-01	5.5E-01	4.1E-01	4.8E-01	5.5E-01	7.9E-01	9.2E-01	1.1E+00
1968	2.3E-01	2.7E-01	3.1E-01	2.3E-01	2.7E-01	3.1E-01	2.6E-01	3.0E-01	3.5E-01
1969	1.7E-01	1.9E-01	2.2E-01	1.7E-01	1.9E-01	2.2E-01	4.2E-01	4.9E-01	5.7E-01
1970	1.8E-01	2.1E-01	2.4E-01	1.8E-02	2.1E-02	2.4E-02	2.6E-01	3.1E-01	3.5E-01
1971	3.4E-02	4.0E-02	4.6E-02	3.4E-02	4.0E-02	4.6E-02	1.5E-04	1.8E-04	2.1E-04
1972	3.5E-02	4.0E-02	4.7E-02	3.5E-02	4.0E-02	4.7E-02	2.6E-01	3.0E-01	3.5E-01
1973	4.2E-02	5.0E-02	5.7E-02	4.2E-02	5.0E-02	5.7E-02	4.2E-01	5.0E-01	5.7E-01
1974	1.7E-02	2.0E-02	2.3E-02	1.7E-02	2.0E-02	2.3E-02	1.7E-01	2.0E-01	2.3E-01
1975	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	2.6E-01	3.1E-01	3.5E-01
1976	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	2.6E-02	3.0E-02	3.5E-02
1977	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	2.5E-02	3.0E-02	3.4E-02
1978	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	3.2E-02	3.8E-02	4.4E-02
1979	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	3.3E-02	3.9E-02	4.5E-02
1980	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	3.4E-02	4.0E-02	4.6E-02
1981	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	3.4E-02	4.0E-02	4.6E-02
1982	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	5.2E-02	6.0E-02	7.0E-02
1983	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	3.2E-03	3.8E-03	4.4E-03
1984	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	4.4E-02	5.2E-02	6.0E-02
1985	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
1986	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
1987	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
1988	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
1989	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
1990	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00
1991	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00	0.0E+00

[1] Note regarding Scientific Notation: 3.2E+02 equals 3.2×10^{2}, or 320; 3.2E-02 equals 3.2×10^{-2}, or 0.032.

Table 5.8 Three most important contributors to the uncertainty in the annual release rates for different time periods.

Time Period	Contributor to Total Uncertainty	Percent Contribution
1947	Ratio, Gross beta to gamma	82
	Grab sampling	13
	Counting efficiency	3
1948	Grab sampling	44
	Flow rate	33
	Counting efficiency	13
1949-52	Grab sampling	44
	Flow rate	34
	Counting efficiency	13
1953-55	Grab sampling	58
	Flow rate	16
	Counting efficiency	13
1956	Flow rate	41
	Grab sampling	37
	Counting efficiency	10
1957-58	Flow rate	40
	Grab sampling	38
	Counting efficiency	11
1959-60	Flow rate	45
	Grab sampling	40
	Random nature of radioactivity	9
1961-62	Flow rate	67
	Random nature of radioactivity	30
	Laboratory processing	3
1963-91	Flow rate	62
	Random nature of radioactivity	35
	Laboratory processing	2

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6.0 ESTIMATION OF HISTORICAL RADIONUCLIDE CONCENTRATIONS IN CLINCH RIVER WATER AND SEDIMENTS

Evaluation of potential health risks from past releases of radionuclides from White Oak Creek requires the estimation of concentrations in environmental media to which people might have been exposed, in particular, historical radionuclide concentrations in Clinch River water and shoreline sediments. Releases to White Oak Creek from the X-10 site [later called Clinton Laboratory and Oak Ridge National Laboratory (ORNL)] started in 1944 after the decision was made to continue operating the laboratory. Environmental measurements of ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co in Clinch River water were used when possible, depending on the availability and quality of the data, for the years 1960 through 1990 (Sections 6.1 - 6.3). Estimates of radionuclide concentrations in Clinch River water obtained with the HEC-6-R model were used for the remaining radionuclides and years (Sections 6.4 - 6.12), based on estimates of discharges to the Clinch River from White Oak Dam (Section 5). Concentrations of radionuclides in shoreline sediments were estimated, using the HEC-6-R model, for all radionuclides and all years (Sections 6.4 - 6.11, 6.13). Radionuclide concentrations were estimated as annual average concentrations in water or sediment at each location of interest (CRM 0, 3.5, 14, and 20.5; see Section 7).

6.1 Historical Measurements of Radionuclide Concentrations in Clinch River Water

For the purposes of this dose reconstruction, actual measured water concentrations were used whenever possible. Based on an evaluation of the nature and sources of uncertainties in these measurements, the measured water concentrations are expected to have smaller uncertainties than do the modeled water concentrations. In particular, the measurements are expected to be more reliable than the model for the years after 1960, when the system became considerably more complex due to the effects of the construction and operation of Melton Hill Dam.

Measured water concentrations for ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co were obtained from annual monitoring reports. Concentrations were reported for the years 1957 to 1990 at Clinch River mile (CRM) 4.5, CRM 14.5, or both (ORNL, 1958; 1959; 1960; 1961; 1962; 1963; 1964; 1965; 1966; 1967; 1968; 1969; 1970; 1971; 1973; 1977; 1978; 1979; 1980; 1981; 1982; 1985; UCC, 1972; 1973; 1974; 1975; 1976; MMES, 1984; 1986; 1987; 1988; 1989; 1990; 1991). At CRM 4.5, the reported annual average concentrations were the averages of quarterly composites of daily grab samples. At CRM 14.5, the reported annual average concentrations were the averages of quarterly composites from a continuous proportional sampler. For most years, concentrations were reported for either CRM 4.5 or CRM 14.5; for 1971-1979, concentrations were reported at both locations (Table 6.1).

Table 6.1 Measured annual average concentrations of radionuclides in the Clinch River at CRM 14.5 and CRM 4.5 (Bq L^{-1})·

	Cs-137		Sr-90		Ru-106		Co-60	
Year	**CRM 14.5[a]**	**CRM 4.5[b]**	**CRM 14.5**	**CRM 4.5**	**CRM 14.5**	**CRM 4.5**	**CRM 14.5**	**CRM 4.5**
10/25/56-11/27/57	-[c]	-0.46	-	#0.97	-	#0.65	-	#0.23
11/27/57-10/23/58	-	-0.27	-	#1.3	-	#0.59	-	-0.17
10/23/58-11/3/59	-	0.20	-	0.69	-	0.42	-	0.085
1960	-	0.085	-	0.35	-	14	-	0.20
1961	-	0.019	-	0.16	-	10	-	0.14
1962	-	0.026	-	0.13	-	5.9	-	0.12
1963	-	0.085	-	0.11	-	3.1	-	0.17
1964	-	0.12	-	0.10	-	1.7	-	0.16
1965	-	0.063	-	0.056	-	0.44	-	0.085
1966	-	0.059	-	0.078	-	0.14	-	0.14
1967	-	0.056	-	0.044	-	0.022	-	0.044
1968	-	0.052	-	0.056	-	0.026	-	0.10
1969	-	0.070	-	0.041	-	0.052	-	0.13
1970	-	0.070	-	0.041	-	0.019	-	0.037
1971	0.048	0.033	0.067	0.037	0.10	0.074	0.022	0.030
1972	0.026	0.019	0.056	0.041	0.030	0.022	-	-
1973	0.019	0.015	0.044	0.063	0.019	0.019	-	-
1974	0.0019	0.0030	0.021	0.017	0.0048	0.011	-	-
1975	0.0026	0.0019	0.016	0.011	0.0048	0.0070	-	-
1976	0.0011	0.00074	0.0096	0.0089	0.0052	0.0056	-	-
1977	0.0011	0.019	0.0067	0.010	0.0056	0.0070	-	-
1978	0.043	0.026	0.0041	0.0037	0.031	0.027	0.0059	0.0041
1979	0.00074	0.00074	0.015	0.012	0.0030	0.0041	0.0019	0.0015
1980	0.0030	-	0.028	-	0.0063	-	0.0078	-

Table 6.1 (continued).

	Cs-137		Sr-90		Ru-106		Co-60	
Year	**CRM 14.5[a]**	**CRM 4.5[b]**	**CRM 14.5**	**CRM 4.5**	**CRM 14.5**	**CRM 4.5**	**CRM 14.5**	**CRM 4.5**
1981	0.0043	-	0.053	-	<0.0010	-	0.0035	-
1982	0.025	-	0.067	-	-	-	0.022	-
1983	0.010	-	0.074	-	-	-	0.0059	-
1984	<0.0089	-	0.041	-	-	-	<0.011	-
1985	0.0063	-	0.070	-	-	-	0.0052	
1986	<0.010	-	0.070[d]	-	-	-	<0.017	-
1987	<0.093	-	0.12[d]	-	-	-	<0.093	-
1988	<0.025	-	0.070[d]	-	-	-	<0.020	-
1989	0.018	-	0.081[d]	-	-	-	0.011	-
1990	0.0059	-	0.031[d]	-	-	-	0.011	-

[a]CRM 14.5 is near Grassy Creek and the K-25 water intake.
[b]CRM 4.5 is near the Kingston Steam Plant (CRM 3.5).
[c]No concentration was reported.
[d]Reported for ^{89}Sr and ^{90}Sr, but expected to be primarily ^{90}Sr.

The first routine monitoring of Clinch River water was conducted in 1957, at CRM 4.5 (Center's Ferry) (ORNL, 1958). The Applied Health Physics report for 1957 states the following:

> "A routine program of determining the fission product contamination in the water of the Clinch River at a point downstream where it becomes available to large population users was started at the beginning of this year. A daily grab sample of approximately a quart is taken from the river at Center's Ferry near Kingston, Tennessee, and composited into a quarterly sample. The composite sample, approximately 20 gallons, is filtered to remove the suspended solids. The filtrate is concentrated by evaporation to a volume of about 1 gallon and the concentrate analyzed for fission products. The suspended solids are weighed and analyzed for fission products."

Sampling at locations upstream from CRM 20.8 was added in late 1959. The upstream sampling location changed several times. Starting in 1966, samples were taken at Melton Hill Dam (CRM 23.1). By 1971, continuous proportional samples were collected at CRM 23.1 and CRM 14.5 [Oak Ridge Gaseous Diffusion Plant (ORGDP) water intake] (UCC, 1972). Daily grab samples continued at CRM 4.5. All Clinch River samples were composited for quarterly analysis (White Oak Dam samples were composited monthly). Sampling procedures did not seem to change much over the years, although improvements in counting technology were incorporated.

For several years (1955-1967), total radioactivity in the Clinch River was reported on a weekly basis. In most cases, the radioactivity was reported as a fraction of the maximum permissible concentration, rather than as an actual concentration. It is not always clear whether this information was based on concentrations calculated at CRM 20.8 or on concentrations measured at CRM 4.5.

6.2 Sources of Uncertainty in Measured Water Concentrations

Several potential sources of uncertainty exist in measured Clinch River water concentrations. These sources of uncertainty and their potential impact on the data are discussed below. Because quantitative estimates of uncertainty were made individually for each of these factors in Section 5, they are not repeated here. However, after consideration of the sources of uncertainty discussed below, multiplicative uncertainty factors were developed to express the combined uncertainty about the measured water concentrations at each location of concern (Section 6.3).

6.2.1 Sampling from Fully vs. Partially Mixed Water

A surface water sample might not have yielded a representative concentration for a target receptor if the station was located in an area of incomplete mixing or at a location not relevant to the target individual. Past tracer studies and calculations by the project team indicate that the water is completely mixed at both sampling locations (CRM 14.5 and CRM 4.5). The relationships of the concentrations at CRM 14.5 and

CRM 4.5 are generally consistent (Table 6.1). Therefore, this source of uncertainty is not considered a significant contributor to the overall uncertainty in the measured concentrations.

6.2.2 Sample Compositing or Averaging Methods

Errors could exist in the measured data if proper averaging techniques were not used. This is especially relevant at CRM 4.5, where grab sampling (rather than continuous, flow-proportional sampling) was performed. Variations in flow and contaminant concentrations can affect the water concentrations obtained via grab sampling techniques. However, for the proportional sampling technique used at CRM 14.5, this source of uncertainty should be relatively low.

As a check on the annual average radionuclide concentrations in the Clinch River reported by ORNL, a comparison was made with results from a two-year study performed by the U.S. Public Health Service (Churchill et al., 1965). Churchill and his colleagues reported the results of weekly samples analyzed by the U.S. Public Health Service between December 1960 and November 1962. For most sampling locations on the Clinch and Tennessee Rivers, daily subsamples were taken; the volumes of the subsamples were selected so as to be proportional to daily stream flows obtained from the USGS; and the subsamples were then composited for weekly analysis. All radiological determinations were made by the U.S. Public Health Service.

Problems with the data reported by Churchill et al. (1965) include ^{137}Cs measurements that were considered to be suspect, due to the high concentrations of ^{106}Ru and an expected high biasing of data for sediment-borne radionuclides as a result of sampler malfunction above Center's Ferry (CRM 5.5) between September 9 and December 2, 1961. The authors state that if they were to repeat the study, they would chemically separate the ^{137}Cs prior to analysis. Churchill et al. (1965) disregarded the data for ^{137}Cs from the period of sampler malfunction at the Center's Ferry site.

Using the weekly results reported by Churchill et al. (1965) with weekly flow measurements, the Task 4 project team estimated annual average concentrations for ^{90}Sr and ^{106}Ru for 1961 and for 11 months of 1962. These radionuclides were not subject to major bias from the sampling problems reported in Churchill et al., as these radionuclides do not interact greatly with the sediment. The difference between the reported ORNL annual averages and the project team's estimated annual averages was a factor of 2 or less, with the exception of ^{137}Cs. Although month-to-month discrepancies occurred, for the entire period of study (December 1960-November 1962), Churchill et al. (1965) report that their calculations for total radionuclide in the river system (total release or total "load") during that period were in good general agreement with ORNL's measurements. ORNL's measurements were lower than Churchill et al.'s by 10-14% for ^{137}Cs, ^{90}Sr, and ^{60}Co (14, 12, and 10%, respectively). For ^{106}Ru, ORNL and Churchill et al. were in very close agreement for the first 12 months, while ORNL was higher by 22% for the second 12 months.

6.2.3 Detector Efficiency

Variations in the efficiency of a radiation detector relative to the efficiency value that is assumed in the calculations underlying the reported data can result in errors in the reported data. Detection methods improved during the period of concern, and the efficiencies of the detectors presumably were incorporated into the reported measurements. In addition, changes in the relative proportions of radionuclides present can lead to changes in the efficiency with which a radiation detector responds to incident radiations. It is not known if the assumed efficiencies were updated appropriately. Section 5.3 presents the quantitative assessment of the uncertainty associated with detector efficiency.

6.2.4 Laboratory Processing

This potential source of uncertainty is discussed in detail for the source term estimates (Section 5.3), but it is also relevant to the measured water concentrations. Analytical techniques performed by different technicians could lead to errors in the reported concentrations.

6.2.5 Stochastic Nature of Radioactive Decay

This is a very minor source of uncertainty, but one that should be mentioned. Slightly different measurements can be obtained if sufficient counting time is not used, due to the stochastic nature of counting radioactive decay. Section 5.3 addresses the quantification of this uncertainty in greater detail.

6.2.6 Detection Limit Effects

Uncertainty associated with measurements of radioactivity can be elevated when ambient levels approach or fall below the limits of detection of the measurement system. Uncertainties in reported values can also be elevated if samples for which the measured radioactivity is below the limits of detection of the counting system are assigned a result equal to the limit of detection. The potential for this error was probably highest in the 1980s, when radionuclide concentrations in Clinch River water were significantly lower than those in the previous years. For some radionuclides, the concentrations reported during these periods were often at or near the detection limit. This could have resulted in overestimation of true concentrations.

Approximate lower limits of detection for Clinch River water samples in the early 1960s were 0.001 Bq L^{-1} (0.03 pCi L^{-1}) for ^{90}Sr and 0.04 - 0.4 Bq L^{-1} (1-11 pCi L^{-1}) for ^{137}Cs, ^{60}Co, and ^{106}Ru (Churchill et al., 1965). The actual limit was dependent on the isotope, the concentration of other isotopes in the same sample, and whether the measurement was for dissolved, suspended, or total contaminant. Measurements known to have included some samples below detection limits (i.e., reported as “less-than” values) were not used in the present analysis.

6.3 Use of Measured Radionuclide Concentrations in Clinch River Water

After consideration of the sources of uncertainty listed in Section 6.2, the project team determined that the reported water data from the ORNL monitoring reports were of sufficient quality to be reasonably representative of the average water concentrations to which certain target individuals would have been exposed. Therefore, when data were available, they were used in preference to model results (Sections 6.4 and following). The measured annual average concentrations of radionuclides in Table 6.1 were used as the basis for further calculations, with the adjustment and uncertainty factors described below. Note that the measurements include radionuclides from all sources (described in detail in Section 6.12); releases from White Oak Creek are the dominant source. Numbers given as "less-than" values (e.g., < 0.0089 Bq L^{-1} for ^{137}Cs in 1984) were not used. In addition, water concentrations for 1957-1959 were not used, as the sampling periods did not quite correspond to the calendar year (e.g., October 1958-November 1959). Starting from the available measurements, annual average water concentrations for 1960-1990 were estimated for the four locations of interest: CRM 0, CRM 3.5, CRM 14, and CRM 20.5.

Based on an evaluation of factors contributing to uncertainty in the measurements (Section 6.2), together with the assessment of uncertainties in the source term estimates (based on related measurements; Section 5.3), uncertainty factors were estimated for application to the measured radionuclide concentrations in Clinch River water. The measurements for CRM 4.5 are expected to be representative for that area within a factor of 1.8; measurements for CRM 14.5 are expected to be representative within a factor of 1.5. These factors are objective estimates of the overall uncertainty associated with the sampling and measurement processes. In the dose and risk calculations, data at CRM 4.5, with an uncertainty factor of 1.8 (expressed as a log-uniform distribution between 0.55 and 1.8), were used directly for assessments for CRM 3.5. Data at CRM 14.5, with an uncertainty factor of 1.5 (expressed as a log-uniform distribution between 0.67 and 1.5), were used directly for assessments for CRM 14.0 (Table 6.2).

Monitoring data were available for only two locations (CRM 14.5 and CRM 4.5); except for 1971-1979, data were available for only 1 location each year. It was therefore necessary to adjust the available measurements and their uncertainty bounds to estimate the concentrations to be used in assessment calculations for other locations. Comparison of the measurements at CRM 4.5 and CRM 14.5 for 1971-1979 (Table 6.1) shows that measured concentrations at CRM 14.5 are generally between a factor of 0.6 to 1.8 times the measured concentration at CRM 4.5 for the same year. The geometric mean of the observed CRM 14.5-to-CRM 4.5 ratios is 1.1 (a single extreme outlier of 0.06--^{137}Cs in 1977--was not included in the calculation). Therefore, an adjustment factor of 1.1 was used to estimate concentrations at CRM 14 from measurements at CRM 4.5 (Table 6.2). In addition, the uncertainty estimate was increased to a factor of 2 on the adjusted concentrations. The adjustment factor and uncertainty factor were combined and expressed as a log-uniform distribution from 0.55 to 2.2 (central value, 1.1; Table 6.2). For estimation of concentrations at CRM 3.5 from measurements at CRM 14, an adjustment factor of 0.91 (1/1.1) was used, and the uncertainty estimate was again increased to a factor of 2 on the adjusted

Table 6.2 Summary of uncertainty factors applied to measured water concentrations.

Location of Measurements	Location Used and/or Adjusted For	Minimum Value	Central Value[a]	Maximum Value	Distribution
CRM 4.5	CRM 3.5	0.55	1.0	1.8	log-uniform
CRM 14.5	CRM 14	0.67	1.0	1.5	log-uniform
CRM 4.5	CRM 14	0.55	1.1	2.2	log-uniform
CRM 14.5	CRM 3.5	0.45	0.91	1.8	log-uniform
CRM 14.5	CRM 20.5	0.4	1.0	2.5	log-uniform
CRM 4.5	CRM 20.5	0.44	1.1	2.75	log-uniform
CRM 4.5	CRM 0	0.4	1.0	2.5	log-uniform
CRM 14.5	CRM 0	0.36	0.91	2.3	log-uniform

[a] The central value of a log-uniform distribution does not have to be specified; it is shown here for information purposes. Values different from 1.0 indicate a bias correction as described in the text.

concentrations. The adjustment factor and uncertainty factor were combined and expressed as a log-uniform distribution from 0.45 to 1.8 (central value, 0.91; Table 6.2).

No water data are available for CRM 20.5 or CRM 0. However, differences between concentrations at CRM 20.5 (assuming complete mixing) and CRM 14.5 or between CRM 4.5 and CRM 0 are thought to be relatively small. Therefore, measurements at CRM 14.5 or adjusted measurements from CRM 4.5 were used for CRM 20.5, and measurements at CRM 4.5 or adjusted measurements from CRM 14.5 were used for CRM 0. In either case, the uncertainty estimate was increased to a factor of 2.5 to account for the additional uncertainty due to the extrapolation between locations. Thus, for use of measurements at CRM 3.5 for CRM 0 or for use of measurements at CRM 14.5 for CRM 20.5, the uncertainty was expressed as a log-uniform distribution from 0.40 to 2.5 (Table 6.2). For use of adjusted measurements from CRM 4.5 for CRM 20.5, the uncertainty factor of 2.5 was combined with the adjustment factor of 1.1 and expressed as a log-uniform distribution from 0.44 to 2.75 (central value, 1.1; Table 6.2); for use of adjusted measurements from CRM 14.5 for CRM 0, the uncertainty factor of 2.5 was combined with the adjustment factor of 0.91 (1/1.1) and expressed as a log-uniform distribution from 0.36 to 2.3 (central value, 0.91; Table 6.2).

6.4 Modeling of Historical Radionuclide Concentrations in Clinch River Water and Sediments

Historical measurements of radionuclide concentrations in Clinch River water were not available for all radionuclides and years of interest. Few measurements of radionuclide concentrations in shoreline sediments were available for any location or any time period. Therefore, the transport, deposition, and scouring of waterborne radionuclides were modeled using HEC-6-R, a modified version of a sediment transport model (HEC-6) developed by the U.S. Army Corps of Engineers (HEC, 1991). The code was modified at Oak Ridge National Laboratory to include radioactive decay, sorption and desorption of contaminants in multiple sediment layers, and specification of a river network with up to 20 inflow points (Rose et al., 1993; Brenkert et al., 1992). HEC-6-R simulates continuous flow, sediment movement, input of sediments from tributaries, and transport of contaminants in water and sediments.

In the early 1990s, three sediment and contaminant transport models were set up for use on the Clinch River/Watts Bar Reservoir system by a team of scientists from Oak Ridge National Laboratory, the Tennessee Valley Authority, the Iowa Institute of Hydraulic Research, and Battelle Pacific Northwest Laboratories (Rose et al., 1993). The model comparison and modification exercise was conducted to establish methods to permit long-term simulation of ^{137}Cs fate in the Clinch River/Watts Bar Reservoir system so that impacts of accidental releases could be defensibly predicted. The HEC-6-R, CHARIMA, and TODAM models were calibrated using TVA's measurements of sediment accumulation and validated based on ^{137}Cs measurements made as part of the Clinch River CERCLA remedial investigation. Each

model was then used to simulate ^{137}Cs fate and transport from 1946 through 1991, and results were compared.[1]

For modeled Clinch River segments, HEC-6-R and CHARIMA predicted total ^{137}Cs inventories that compared well with estimated Clinch River inventories based on core sample data.[2] TODAM predicted elevated levels. For Watts Bar Reservoir, TODAM and HEC-6-R predicated the patterns of ^{137}Cs deposition well, while all three models predicted lower total ^{137}Cs inventories than indicated by extrapolation between core sample data.

The HEC-6-R model was selected for use on the Task 4 assessment of radionuclide movement in Clinch River as a result of releases from White Oak Creek for the following reasons.

- It was set up and successfully tested for the Clinch River/Watts Bar Reservoir system.
- All historical water discharges were available from all dams and tributaries, and water level elevation data were available for Watts Bar Dam, the downstream boundary[3].
- Sediment influxes had been successfully tested and had resulted in the appropriate sediment deposition and erosion patterns.
- HEC-6-R contaminant transport calculations and results can easily be verified for mass balance (in contrast to the other two models).
- Contaminant transport and trapping agreed well with historical measurements.
- HEC-6-R was the only model among the three that kept track of shoreline contamination.

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[1] Personal communication between Tom Widner of the project team and Dr. Antoinette Brenkert of ORNL, technical advisor to the project team on surface water modeling. April 1997.

[2] Brenkert et al., draft publication in preparation. A. Brenkert, L. Ewing, Y. Onishi, K. Rose, F. Holly, G. Schohl, W. Perkins, and R. Cook. "Simulating Sediment Transport and Contaminant Fate in a Large River/Reservoir System Using Multiple Models: I. Model Descriptions, Calibration and Validation."

[3] Water discharge and elevation data were obtained from Gerald Schohl of TVA.

6.5 Description of the HEC-6-R Model

HEC-6-R is a one-dimensional numerical model that approximates a continuous water discharge by a sequence of steady flow discharges. A summary of some key features of the model is provided in this section and in Figure 6.1. Additional details can be obtained from the original documentation of the model (HEC, 1991).

The HEC-6-R model predicts daily contaminant concentrations in the following components of a surface water system:

- the dissolved phase in which soluble contaminants are transported along the flowing stream;
- river bed sediment that deposits on the bottom of the stream and can be removed by scouring;
- river bank (shoreline) soil that is contaminated when the stream periodically inundates its banks; and
- the suspended phase in which contaminants are in the form of particles or attach to particles that remain suspended in the flowing stream.

HEC-6-R determines water surface profiles by backwater calculations using a standard step method to solve the energy and continuity equations (HEC, 1991). Water velocity, depth, width, slope, and bed shear stress are calculated from the water surface profile calculations.

HEC-6-R uses ten sediment size classes in simulating transport of particles in the flowing water, deposition into the sediment layers, and detachment allowing further movement (erosion). The size classes are also categorized as clays or silts (which are cohesive) or sands (which are non-cohesive).

HEC-6-R recognizes an active sediment layer continuously mixed by the flowing water, and an inactive layer that serves as a source or sink for the active layer. In addition, multiple inactive sublayers function sequentially as sources or sinks for the inactive sublayer. The volumes and thicknesses of the active and inactive sediment layers are evaluated at each time step of the modeling. The model simulates a slow moving surface layer that shields finer particles from being entrained in the flowing stream, in a process called armoring.

HEC-6-R reflects the flooding of shoreline sediments when high water conditions occur. Shoreline sediments can undergo the same sedimentation, erosion, and contaminant sorption and desorption as river bed sediments. By keeping track of the levels of contamination left on the shore during high flow events, shoreline sediment concentrations are predicted as a function of time.

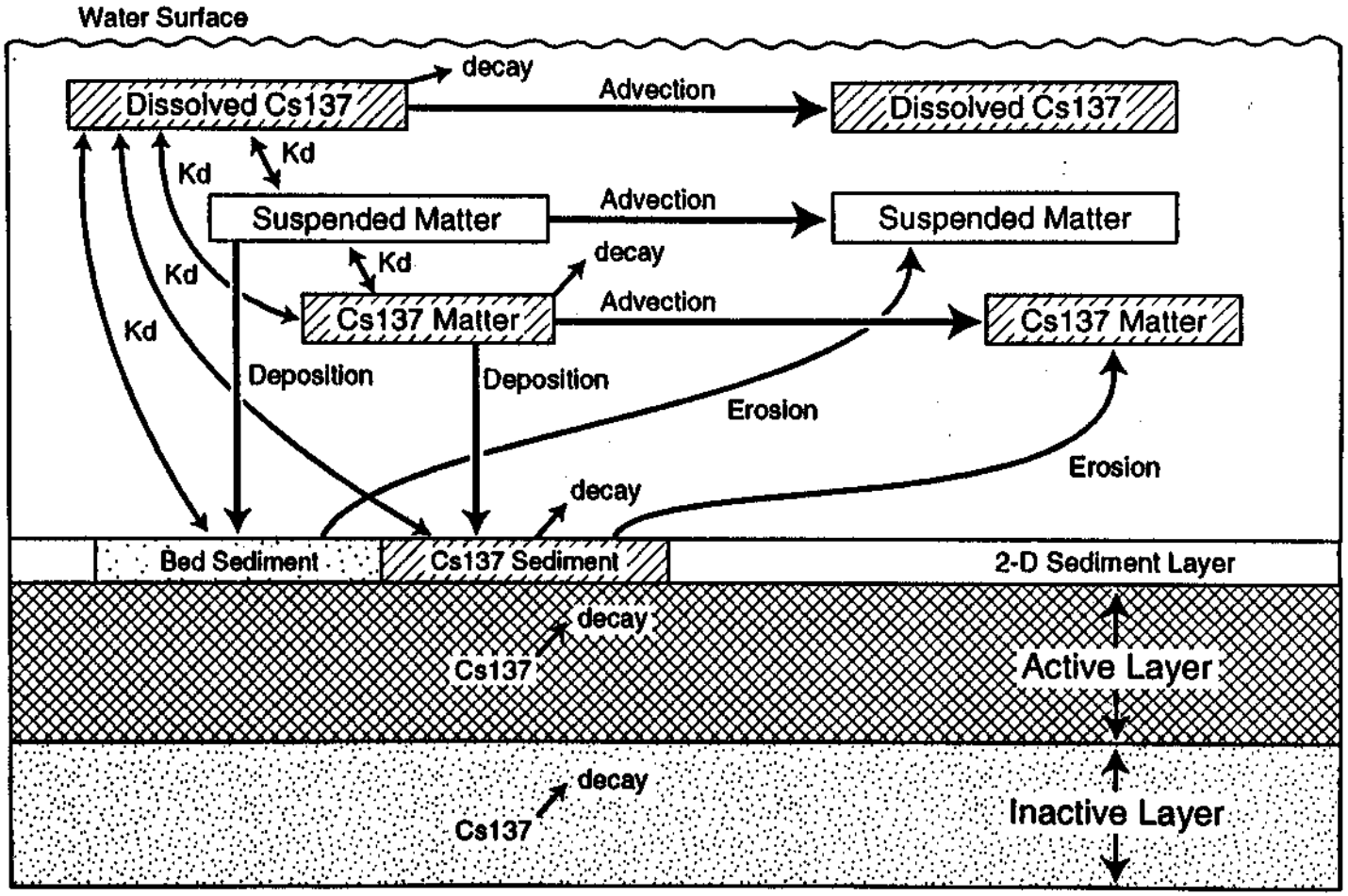

Figure 6.1 Schematic of the processes modeled by HEC-6-R (from Clapp et al., 1996).

Rates of sorption and desorption of contaminants are determined by their distribution coefficients (K_ds) that are specific for the contaminant and the sediment particle sizes and types. Distribution coefficient is defined as the ratio, at equilibrium of contaminant concentration in water to the contaminant concentration adsorbed on sediment particles. Suspended sediment is assumed to be fully mixed throughout the volume of water moving from one section of the river to the next.

6.6 Characterization of the Clinch River for Modeling Purposes

In order to simulate the fate of contaminants released to a flowing body of water, that river or stream must be described in terms of its geometry, water flow rates and depths, and sediment particle sizes and distribution patterns. For application of the HEC-6-R model, the Clinch River and its main tributaries were characterized as described below[1].

6.6.1 Channel Bed Geometry

Channel bed geometry was specified at 19 cross sections on the Clinch River, based on measurements made by TVA in 1946. Cross sections were specified in terms of coordinates of "stations" with associated elevations. Distances along the river between these cross sections were also specified. Another set of 37 cross sections was specified for Poplar Creek based on TVA measurements, and a set of 6 cross sections was specified for White Oak Creek Embayment based on topographic maps and 1991 measurements at its mouth.

6.6.2 Water Discharges from Upstream Inputs

Water flow rates from the major inputs to the Clinch River/Watts Bar Reservoir system were specified as a function of time. These included the following.

- Fort Loudoun Dam;
- Melton Hill Dam (after it was completed in December 1962);
- Clinch River near Clinch River Mile 22.3 (CRM 22.3) before 1963;
- Little Tennessee River (through 1976) and Emory River;
- Poplar Creek, White Oak Creek, and other free-flowing streams (White, Piney, Sweet, Pond, Paint, Riley, Caney, and King).

[1] Personal communication between Tom Widner of the project team and Dr. Antoinette Brenkert of ORNL, technical advisor to the project team on surface water modeling. April 1997.

Complete (hourly or daily) discharge data for TVA's dams and several tributaries of the Clinch River/Watts Bar Reservoir system for the period of interest were obtained from TVA[1] and USGS records (USGS, 1953-1955; 1961-1964; 1993). However, only partial data were available for Poplar Creek and White Oak Creek, and no measurements were available for most of the small tributary streams. For tributaries with no measurements, flow rates were estimated based on measurements from similar tributaries that were adjusted according to the ratio of their respective drainage or "catchment" areas. This process of estimating flow rates based on similar streams is called "scaling." For streams larger than Poplar Creek, discharges were scaled based on Emory River measurements. For streams smaller than Poplar Creek, discharges were scaled based on the Poplar Creek measurements.

Flow measurements for White Oak Creek were recorded by the USGS during the 1950s and 1960s. However, between 1944 and 1952, no documented USGS flow data were available. As described in Section 5, available measurements from three gauging stations on two similar surface water systems were scaled according to the sizes of their drainage areas to estimate the flow over White Oak Dam. The method used for this scaling and the sources of data used are described in Section 5.3.2. Based on analysis of measured and scaled White Oak Creek discharges for periods for which both were available, scaled Emory River flow measurements were selected by the project team for use in predicting White Oak Dam flow rates during November through April of each year, and scaled measurements from the Zion gauging station on the Little Chestuee River were used for predicting White Oak Creek discharges for May through October.

6.6.3 Sediment Loads and Particle Sizes in Upstream Inputs

Quantities of sediment flowing from upstream inputs were estimated using available sediment rating curves for tributaries. These rating curves predict sediment flow rates in a stream for given water flow rates. For example, following is the equation of a rating curve for Poplar Creek near Oak Ridge (Ewing, 1993):

$$q_s = 0.686\ q_w^{1.74}$$

Where

q_s = sediment flow rate (ton d^{-1} mi^{-2}) and
q_w = water flow rate (ft^3 sec^{-1} mi^{-2}).

Rating curves are also available for the Emory River at Oakdale, Tennessee, the Little Tennessee River at McGhee, Tennessee, and White Oak Creek at White Oak Dam[2] (Ewing, 1993; Gaydos et al., 1982).

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[1]Personal communication between Dr. Antoinette Brenkert and Gerald Schohl of TVA.

[2]Personal communication between Antoinette Brenkert, technical advisor to the project team, and Thomas Fontaine of Oak Ridge National Laboratory (1993).

6.6.4 Stage Heights at Watts Bar Dam

In addition to water discharges from upstream inputs to the Clinch River/Watts Bar Reservoir system, stage height values (water elevations) at Watts Bar Dam were also specified for use in HEC-6-R modeling. These values were obtained from TVA for the time period of interest[1].

6.6.5 Particle Size Distributions in the Clinch River

The particle size distribution used in the Task 4 modeling of the Clinch River/Watts Bar Reservoir system was determined from a number of historical sources of information.

- The particle size distribution of two cores taken at CRM 7.5 and CRM 14 indicated 14.2% sand, 63.3% silt, and 22.5% clay (Struxness et al., 1967).

- The mean particle size distribution of composite samples of 45 Clinch River sediment cores was 23% sand, 54% silt, and 23% clay (Carrigan et al., 1967).

- The mean particle size distribution reported for the Clinch River by TVA (1986), based on a variable number of samples, was 34-49% sand, 66-69% silt, and 46% clay.

- The mean particle sizes reported for the Tennessee River by TVA (1986), based on a variable number of samples, were 64%-88% silt and clay, with the remainder being sand.

- The mean particle size distribution reported for Poplar Creek by TVA (1986), based on three samples, was 24% sand, 45% silt, and 31% clay.

- A sediment-type map of lower Watts Bar Reservoir (Olsen et al., 1992).

Tables 6.3, 6.4, and 6.5 present sediment particle-size distributions and transport parameters that were used in the sediment fate simulations. In specification of particle-size distributions, dams were assumed to cause trapping of coarser materials upstream. Sediment sizes in White Oak Creek were adjusted to best match deposition patterns: coarser sediments initially deposited in large quantities in White Oak Creek Embayment, and finer sediments initially left the system over Watts Bar Dam in excessive quantities. For the beginning of the simulation in 1944, the initial bed sediments were assumed to be rather coarse because of the riverine characteristics of the total system.

By 1990, as predicted in HEC-6-R modeling, the Watts Bar Reservoir bed sediment was, in fact, found by sampling to consist mainly of mud made up of clay and silt.

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[1]Personal communication between Antoinette Brenkert and Gerald Schohl of TVA.

Table 6.3 Sediment particle size distributions used in HEC-6-R modeling.

Fractional Composition of Sediments

Sediment Type	Loudoun Dam	Melton Hill Dam		White, Piney, Emory, Poplar, Other Minor Local Streams	White Oak Creek	Initial Bed Composition Available for Redistribution	Particle Size[1] (Geometric Mean)
		Before 1963	After 1963				
Clay	.70	.62	.70	.62	.35	.22	.0027
Very Fine Silt	.14	.10	.14	.10	.20	.18	.0056
Fine Silt	.10	.08	.10	.08	.18	.16	.0110
Medium Silt	.04	.05	.04	.05	.12	.14	.0220
Coarse Silt	.02	.05	.02	.05	.10	.12	.0440
Very Fine Sand	0	.05	0	.05	.03	.08	.0880
Fine Sand	0	.03	0	.03	.02	.04	.1770
Medium Sand	0	.02	0	.02	0	.02	.3540
Coarse Sand	0	0	0	0	0	.02	.7070
Very Coarse Sand	0	0	0	0	0	.02	1.414

[1] Size classes follow the American Geophysical Union Classification Scale.

Table 6.4 Sediment transport parameters used in HEC-6-R sediment fate simulation.

Sediment Type	**Geometric Mean Particle Diameter[1] (mm)**	**Fall Velocity[2] (cm s^{-1})**	**Density[3] (g cm^{-3})**	**Compaction Rate (g cm^{-3} y^{-1})**	***In Situ* Dry Density (g cm^{-3})**
Clay	0.0027	0.000642	2.61	0.171	0.529– 0.737
Silt	0.0056	0.00256	2.61	0.043	1.041– 1.185
	0.0110	0.0102	2.61	0.043	1.49
	0.0220	0.0402	2.61	0.043	1.49
	0.0440	0.156	2.65	0.043	1.49
Sand	0.0880	0.582	2.65	Not Applicable	1.49
	0.1770	1.803	2.65		1.49
	0.3540	4.131	2.65		1.49
	0.7070	8.632	2.65		1.49
	1.4140	14.705	2.65		1.49

[1] Size classes follow the American Geophysical Union Classification Scale.
[2] Calculated following the method of Williams (1980).
[3] References: Simons and Senturk, 1977; USACE, 1993.

Table 6.5 Other sediment transport parameters for silt and clay.

Parameter	Value
Critical Shear for Clay Deposition (dyne cm^{-2})	32.6
Critical Shear for Silt Deposition (dyne cm^{-2})	11.5
Critical Shear for Clay Erosion (dyne cm^{-2})	42.6
Critical Shear for Silt Erosion (dyne cm^{-2})	14.3
Surface and Mass Erosion Rate (g cm^{-2} s^{-1})	shear stress dependent

References: Struxness et al., 1967; personal communication between A. Brenkert and R.B. Krone of the Hydraulic Engineering Laboratory, University of California, Berkeley.

6.7 Specification of Radionuclide Releases at White Oak Dam

As discussed in Section 5, estimates of the quantities of each radionuclide of interest were specified by the project team in the form of annual central estimates of curies released along with a 95% subjective confidence interval. Historical records located by the project team were not sufficient to support development of release estimates on a finer time resolution. The annual release totals were converted to daily release quantities because a daily time step was needed in HEC-6-R modeling so that regions of deposition and erosion would be well represented and reported accurately as a function of time. The annual release quantities were converted to average daily releases for the period of interest based on the fraction that the day's water discharge at White Oak Dam contributed to the total water discharge for the year at White Oak Dam, using the following formula:

$$\textit{Daily Release } (Bq\ d^{\&1}) \text{ ' } \textit{Annual Release } (Bq) \times \frac{\textit{Daily Water Discharge } (m^3\ d^{\&1})}{\textit{Total Annual Water Discharge } (m^3)}$$

6.8 Specification of Contaminant Behavior

The behavior of specific radionuclides in a surface water system depends to a great extent on the solubility of the physical and chemical forms of the contaminant that are present and the extent to which they become associated with particles of various sizes. In general terms, ^{106}Ru and ^{131}I are highly soluble and have been found to move rapidly through river systems. On the other hand, ^{137}Cs, ^{60}Co, ^{95}Zr, ^{95}Nb, and ^{144}Ce are often called "particle reactive" because they adsorb to sediments quite readily. While adsorption and desorption of particle-reactive contaminants to and from sediments is considered a dynamic process, distribution coefficients (K_ds) are commonly reported; these values represent levels of contaminants

adsorbed relative to the levels in solution at equilibrium. This is primarily because the equilibrium is established very quickly.

For example, Olsen et al. (1992) measured K_ds for ^{137}Cs for suspended matter at Clinch River Mile 0.2 and in lower Watts Bar Reservoir around Tennessee River Mile 555; values were 2×10^5 and 3×10^5, respectively. K_d values for ^{137}Cs for river bed sediments reported from other systems include 5.2×10^3, 2.7×10^4, 3.2×10^4, 500 to 5.6×10^4, and for suspended clay from 2.7×10^4 to 1.4×10^5, suspended silt from 1×10^3 to 9.9×10^4, and suspended sand from 400 to 1.5×10^4 in Ukraine's Dnieper Reservoirs after the Chernobyl accident (Yu et al., 1993; Jirka et al., 1983; Personal communication between A. Brenkert and Y. Onishi of Battelle Pacific Northwest Laboratories).

For ^{137}Cs, ^{90}Sr, ^{60}Co, and ^{106}Ru, the K_d values were adjusted to calibrate the HEC-6-R model using estimates of the sediment inventories of these four radionuclides based on historical sediment core data. In a series of preliminary HEC-6-R model runs, the K_d values for these radionuclides were adjusted so that sediment core inventories predicted by HEC-6-R matched those estimated for the Clinch River in 1962 and 1977, based on a Clinch River inventory of radionuclides.

For ^{95}Nb, ^{95}Zr, ^{131}I, and ^{144}Ce, sediment core data were not available in sufficient number to support a similar calibration process. For these radionuclides, ranges of K_d values were determined from the literature (Yu et al., 1993; Jirka et al., 1983). As shown in Table 6.6, the higher reported K_d values for each radionuclide were assigned to the finer particles (e.g., clays) and the lower values to coarser particles (e.g., sands).

As described in Section 6.10, three model runs were conducted for each radionuclide. For ^{95}Nb, ^{95}Zr, and ^{144}Ce, K_d values from the low end of each nuclide's range were used with the lower-bound source term for one run to estimate a low-end sediment concentration; intermediate K_d values with the central estimate source term for a second run to estimate a central value sediment concentration; and maximum K_d values with the upper-bound source term for a third run to estimate a high-end sediment concentration. The reason for combining the upper-end values of K_d with upper-bound source term values is that all these nuclides (^{95}Nb, ^{95}Zr, and ^{144}Ce) are important only for the external radiation exposure pathway; therefore, this combination would lead to upper-bound concentrations in the sediments. For ^{131}I, the pathway of concern is direct ingestion via drinking water; therefore, the combination of high source term values with low K_d values was used to obtain upper-bound concentrations in the water, and the combination of low source term values with high K_d values was used to obtain lower-bound values. The distribution coefficients used in the assessment of radionuclides released from White Oak Creek are given in Table 6.6.

Table 6.6 Water/sediment distribution coefficient (K_d) values used in the Task 4 assessment (Bq kg^{-1} of sediment per Bq L^{-1} of water, or L kg^{-1}).

Radionuclide	Half-life (d)	K_d for Clay	K_d for Silt	K_d for Sand
^{137}Cs	10,987	30,000	30,000	300
^{90}Sr	10,111	84	9	0.2
^{60}Co	1922	9,300	2,800	51
^{106}Ru	368	300	140	0
^{95}Nb[a]	35	900[b]	550[b]	160[b]
		550-2,000[c]	160-900[c]	100-550[c]
^{95}Zr[a]	65.5	7,300[b]	1,000[b]	600[b]
		1,000-10,000[c]	600-7300[c]	400-1,000[c]
^{131}I[a]	8.04	1[b]	1[b]	1[b]
		0-25[c]	0-25[c]	0-25[c]
^{144}Ce[a]	284	70,000[b]	10,000[b]	7,800[b]
		10,000-140,000[c]	7,800-70,000[c]	5,000-10,000[c]

[a] Available historical environmental measurements were insufficient in number to support site-specific determination of distribution coefficients to match environmental data.
[b] Central value.
[c] Lower and upper bounds.

6.9 Calibration of the Model with Historical Measurements

Various aspects of HEC-6-R model performance were "calibrated" with historical estimates of contaminant inventories in sediments derived from sediment core data so that model results would match the available environmental measurements as closely as possible, and the validity of predictions for other locations and time periods would be established. The usage of site-specific data in preparing the HEC-6-R model for use of the project is described below.[1]

[1] Personal communication between Tom Widner of the project team and Dr. Antoinette Brenkert of ORNL, technical advisor to the project team on surface water modeling. April 1997.

6.9.1 Calibration of Hydrologic Calculations

HEC-6-R's water flow calculations were calibrated by doing parallel runs with another hydrodynamic model, CHARIMA (Holly et al., 1990). CHARIMA was calibrated and validated for hourly flows in the Clinch River/Watts Bar system.

6.9.2 Calibration of Sediment Transport

Sediment transport calibration was achieved by adjusting inflowing sediment types (relative proportions of clay, silt, and sand) and adjusting sediment amounts from the different inflow points to best match historical sedimentation measurements made by TVA. Table 6.7 shows predicted and measured net sediment accumulation in the Clinch River and Watts Bar Reservoir as an indicator of the effectiveness of this calibration.

Table 6.7 Net sediment accumulation in the Clinch River and Lower Watts Bar Reservoir (acre feet)

Time Period and Area	Predicted	Estimated from Measurements	Ratio of Predicted to Estimated
1946–1951 Clinch River	670	690	0.97
1946–1951 Lower Watts Bar Reservoir	2300	2000	1.2
1951–1956 Clinch River	1700	1600	1.1
1951–1956 Lower Watts Bar Reservoir	4900	3000	1.6
1956–1961 Clinch River	610	650	0.94
1956–1961 Lower Watts Bar Reservoir	8300	7800	1.1
1961–1991 Clinch River	62	25	2.5
1961–1991 Lower Watts Bar Reservoir	9800	8900	1.1

6.9.3 Calibration of Contaminant Fate

Contaminant fate was calibrated against independent estimates of sediment inventories in the sediment bed by adjusting the contaminant distribution factors (K_d values) so that predicted core inventories matched measured contaminant inventories in river-bed sediments. As discussed earlier, the calibration was conducted only for ^{137}Cs, ^{90}Sr, ^{60}Co, and ^{106}Ru, because sufficient inventory data were not available for the other radionuclides. This calibration was performed in an iterative fashion during preliminary model runs, during which the contaminant distribution factors for the four primary radionuclides were the only parameters that were changed.

6.9.4 Evaluation of Site-Specific Environmental Data

Inventories of radionuclides in the bed sediments of the Clinch River system (primarily the Clinch River from CRM 22.6 to CRM 0.0 and Watts Bar Reservoir) were compiled from all available environmental data from the 1940s to the present. Sediment inventories were developed for ^{90}Sr, ^{137}Cs, ^{60}Co, and ^{106}Ru. Because of the lack of better data from the 1940s and 1950s, some information was also evaluated for less specific values such as gross beta radioactivity.

The following documents were used to generate contaminant inventories: ORNL Applied Health Physics Reports, 1958 thru 1982; Setter and Kochtitsky, 1950; Cottrell, 1959; Morton, 1962; Churchill et al., 1965; Carrigan and Pickering, 1967; Struxness et al., 1967; Oakes et al., 1982; TVA, 1985; Olsen et al., 1992; and Cook et al.,1992. More recent data (late 1980s and early 1990s) were obtained from the Oak Ridge Environmental Information System database.

The most comprehensive sources of environmental data were the Applied Health Physics reports (annual, quarterly, monthly), the Clinch River Study reports from the early 1960s, the sediment studies from the late 1970s (as detailed in Oakes et al., 1982) and the Remedial Investigation/Feasibility Study work from the late 1980s. Members of the project team reviewed the methods and results of a study of sediment cores collected from the Clinch river in the summer of 1962 (Carrigan and Pickering, 1967; Carrigan et al., 1967; Morton, 1965). Based upon the data provided, upper and lower bounds were estimated for the 1962 inventories (curies) of ^{137}Cs, ^{90}Sr, ^{60}Co, and ^{106}Ru present in Clinch River sediment. Best estimates had been provided in the original reports. The calibration of contaminant fate was performed such that the modeled inventories of the four primary radionuclides matched the best estimates from the 1962 study. Estimated Clinch River sediment inventories for 1977/1978 as reported by Oakes et al. (1982) were then used as a secondary check of the calibration of contaminant fate. The estimated contaminant inventories for 1962 and 1977 are presented in Table 6.8.

6.9.5 Comparison of Estimated Sediment Inventories with Predictions

Estimates of ^{137}Cs, ^{90}Sr, ^{60}Co, and ^{106}Ru in sediment from 1962 and 1977 were used in the calibration of the HEC-6-R model. Table 6.8 presents the predicted sediment inventories from the HEC-6-R model for 1962 and 1977 along with the estimated sediment inventories for these years from actual measurements.

It was possible to match predicted radionuclide inventories with those estimated from 1962 measurements for each of the four primary radionuclides. Agreement of upper and lower bound values from the model and from measurements was quite good for ^{90}Sr, ^{60}Co, and ^{106}Ru, but not as good for ^{137}Cs. Since a rigorous uncertainty analysis was not included in the reports of the 1962 study, and detailed supporting documentation could not be found, it is possible that the uncertainty of the ^{137}Cs inventory estimated from these measurements was under represented. It is also possible

Table 6.8 Measured and predicted radionuclide inventories in the Clinch River for 1962 and 1977.

Radionuclide	Range	Estimated[1] 1962 Inventory (Ci)	HEC-6-R Predicted 1962 Inventory (Ci)	Estimated[2] 1977/1978 Inventory (Ci)	HEC-6-R Predicted 1977/1978 Inventory (Ci)
Cs-137	Upper Bound	180	260	110	170
	Best Estimate	150	150		97
	Lower Bound	120	84		53
Sr-90	Upper Bound	3.4	4.3	2	2.8
	Best Estimate	2.9	2.9		1.9
	Lower Bound	2.4	2.0		1.3
Co-60	Upper Bound	22	25	5	3.1
	Best Estimate	18	18		2.3
	Lower Bound	14	13		1.6
Ru-106	Upper Bound	21	19	Not Available	0.0014
	Best Estimate	16	16		0.0012
	Lower Bound	11	13		0.0010

[1]Developed by the project team based on information in Carrigan and Pickering, 1967; Morton, 1965; and Carrigan et al., 1967.
[2]Oakes et al. 1982.

that the uncertainty in the source term for ^{137}Cs was over estimated. Releases of this key radionuclide were complicated by significant discharges that occurred in 1956 due to scouring of the bed of White Oak Lake, which was drained in late 1955. Releases from this period could have significantly impacted concentrations measured in 1962 and inventories that were estimated from those data.

6.10 Structure of the Modeling Assessment

Waterborne transport in the Clinch River/Watts Bar Reservoir system from 1943 through 1991 was modeled for ^{137}Cs, ^{90}Sr, ^{60}Co, ^{106}Ru, ^{95}Zr, ^{95}Nb, ^{131}I, and ^{144}Ce. In order to reflect uncertainty in the estimates of quantities released from White Oak Dam (all radionuclides) and in the K_d values for radionuclides for which fewer inventory data were available for calibration (^{95}Zr, ^{95}Nb, ^{131}I, and ^{144}Ce), 24 HEC-6-R runs were conducted. A lower bound, a central value, and an upper bound (2.5th, 50th, and 97.5th percentiles) were specified for the release quantity for each radionuclide (Section 5); a HEC-6-R

run was conducted for each of these release values. For ^{95}Nb, ^{95}Zr, and ^{144}Ce, K_d values from the low, intermediate, and high portions of the ranges identified in the literature for each radionuclide were assigned for the lower bound, best estimate, and upper bound model runs, respectively, for estimating concentrations in sediments (see Table 6.6). For ^{131}I, K_d values from the low, intermediate, and high portions of the ranges identified in the literature were assigned for the upper bound, best estimate, and lower bound model runs for estimating concentrations in water. The HEC-6-R model runs are summarized in Table 6.9.

Each modeling run took from 8 to 37 hours to complete, depending on the computer system in use. Runs were completed in McLaren/Hart-ChemRisk offices in Alameda, California, and Cleveland, Ohio, as well as independently at *SENES'* office in Oak Ridge for quality assurance purposes. Results were compared and in all cases were essentially identical.

The HEC-6-R model predicted dissolved, suspended, and sediment/soil concentrations of each radionuclide at the following locations, which are representative of particular portions or "reaches" of the Clinch River:

- At CRM 20.5 . . . . . Near Jones Island
- At CRM 19.7 . . . . . Near Jones Island
- At CRM 19.2 . . . . . Just below Jones Island
- At CRM 14.0 . . . . . Near Grassy Creek (just above the mouth of Poplar Creek)
- At CRM 7.6 . . . . . Below the mouth of Poplar Creek
- At CRM 3.5 . . . . . Near the Kingston Steam Plant
- At CRM 2.6 . . . . . Near the town of Kingston
- At CRM 0.0 . . . . . At the mouth of the Clinch River
- At TRM 529 . . . . . On the Tennessee River near Watts Bar Dam

Locations were selected so as to be spaced at a sufficient number of distances along the reach of interest on the Clinch River that changes in estimated concentrations with increasing distance would be adequately characterized. Locations were specified near the point where White Oak Creek enters the Clinch (near Jones Island), near where several other contributing surface water bodies enter the Clinch (Grassy Creek and Poplar Creek), near an important population center (Kingston), a readily recognizable landmark (the Kingston Steam Plant), and at the mouth of the Clinch River. Modeling was also performed for a location on the Tennessee River near Watts Bar Dam, about 39 miles downstream of the mouth of the Clinch. Of the nine locations addressed in the modeling, four were selected as reference locations for detailed calculations of dose and risk as outlined in the sections of this report that follow. These reference locations correspond to Clinch River Miles 20.5, 14, 3.5, and 0.

Table 6.9 Descriptions of the HEC-6-R model runs.

Run No.	Radionuclide	Release Quantity Value Used	K_d Values Used
1	^{137}Cs	Lower Bound	"Calibrated" with site-specific inventory estimates.
2	^{137}Cs	Central Value	"Calibrated" with site-specific inventory estimates.
3	^{137}Cs	Upper Bound	"Calibrated" with site-specific inventory estimates.
4	^{90}Sr	Lower Bound	"Calibrated" with site-specific inventory estimates.
5	^{90}Sr	Central Value	"Calibrated" with site-specific inventory estimates.
6	^{90}Sr	Upper Bound	"Calibrated" with site-specific inventory estimates.
7	^{60}Co	Lower Bound	"Calibrated" with site-specific inventory estimates.
8	^{60}Co	Central Value	"Calibrated" with site-specific inventory estimates.
9	^{60}Co	Upper Bound	"Calibrated" with site-specific inventory estimates.
10	^{106}Ru	Lower Bound	"Calibrated" with site-specific inventory estimates.
11	^{106}Ru	Central Value	"Calibrated" with site-specific inventory estimates.
12	^{106}Ru	Upper Bound	"Calibrated" with site-specific inventory estimates.
13	^{95}Zr	Lower Bound	From lower ends of literature ranges.
14	^{95}Zr	Central Value	From middle portions of literature ranges.
15	^{95}Zr	Upper Bound	From upper ends of literature ranges.
16	^{95}Nb	Lower Bound	From lower ends of literature ranges.
17	^{95}Nb	Central Value	From middle portions of literature ranges.
18	^{95}Nb	Upper Bound	From upper ends of literature ranges.
19	^{131}I	Lower Bound	From upper ends of literature ranges.
20	^{131}I	Central Value	From middle portions of literature ranges.
21	^{131}I	Upper Bound	From lower ends of literature ranges.
22	^{144}Ce	Lower Bound	From lower ends of literature ranges.
23	^{144}Ce	Central Value	From middle portions of literature ranges.
24	^{144}Ce	Upper Bound	From upper ends of literature ranges.

6.11 Results of Surface Water and Shoreline Sediment Modeling

Comparisons of the predicted annual average radionuclide concentrations in water at CRM 14 and CRM 3.5 with the available measurements (Table 6.1) at CRM 14.5 and CRM 4.5, respectively, are shown in Figures 6.2 through 6.5 for ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co. Sample results of the predicted concentrations of ^{131}I in water and of ^{137}Cs, ^{90}Sr, ^{106}Ru, ^{60}Co, ^{144}Ce, ^{95}Nb, and ^{95}Zr in shoreline sediment are shown in Figures 6.6 through 6.13 for CRM 14. The figures illustrate the time course of the predicted (or predicted and measured) annual average concentrations at the specified locations. The model predictions for all radionuclides are tabulated for the four locations of main interest (CRM 20.5, 14, 3.5, and 0) in Appendix 6A. The uncertainties (95% confidence intervals) on the model predictions in Figures 6.2 though 6.13 and in Appendix 6A reflect only the uncertainties in the source term (all radionuclides) and the K_d values (^{131}I, ^{144}Ce, ^{95}Nb, and ^{95}Zr).

6.12 Sources of Uncertainty in Modeled Water Concentrations

Discrepancies in the measured and modeled concentrations of radionuclides in water (Figs. 6.2-6.5) may be attributable to any of several explanations: For example, the modeled and measured concentrations compared in Figs. 6.2-6.5 are 0.5 or 1 mile apart in location and therefore may reflect differences in localized situations, and the measurements for 1957-1959 do not reflect a standard calendar year. Use of annual source term estimates with daily flows could have produced inaccuracies, even for time-integrated endpoints. In addition, the figures do not indicate the uncertainties in the measured concentrations (Section 6.2), and the uncertainties shown in these figures for the modeled concentrations do not include all known sources of uncertainty.

Other conditions, such as scouring or flooding events not accounted for by the model in certain years (e.g., 1977), could also have resulted in deviations between the modeled and measured concentrations. This source of uncertainty is applicable primarily to particle reactive contaminants. Unknown or unpredicted scouring events or floods could have resulted in resuspension of sediment- sorbed radionuclides into the water column, resulting in higher water concentrations. These events would have been reflected in measurement data. From Table 6.1, it appears that such events probably did occur. For example, the higher measured concentration of ^{137}Cs at CRM 4.5 than at CRM 14.5 in 1977 may have been due to a scouring event between the two sampling locations (perhaps caused by the flooding that occurred in April 1977). Discrepancies between measured and modeled concentrations due to this source of uncertainty alone could be as much as 25 to 50% for some of the radionuclides.

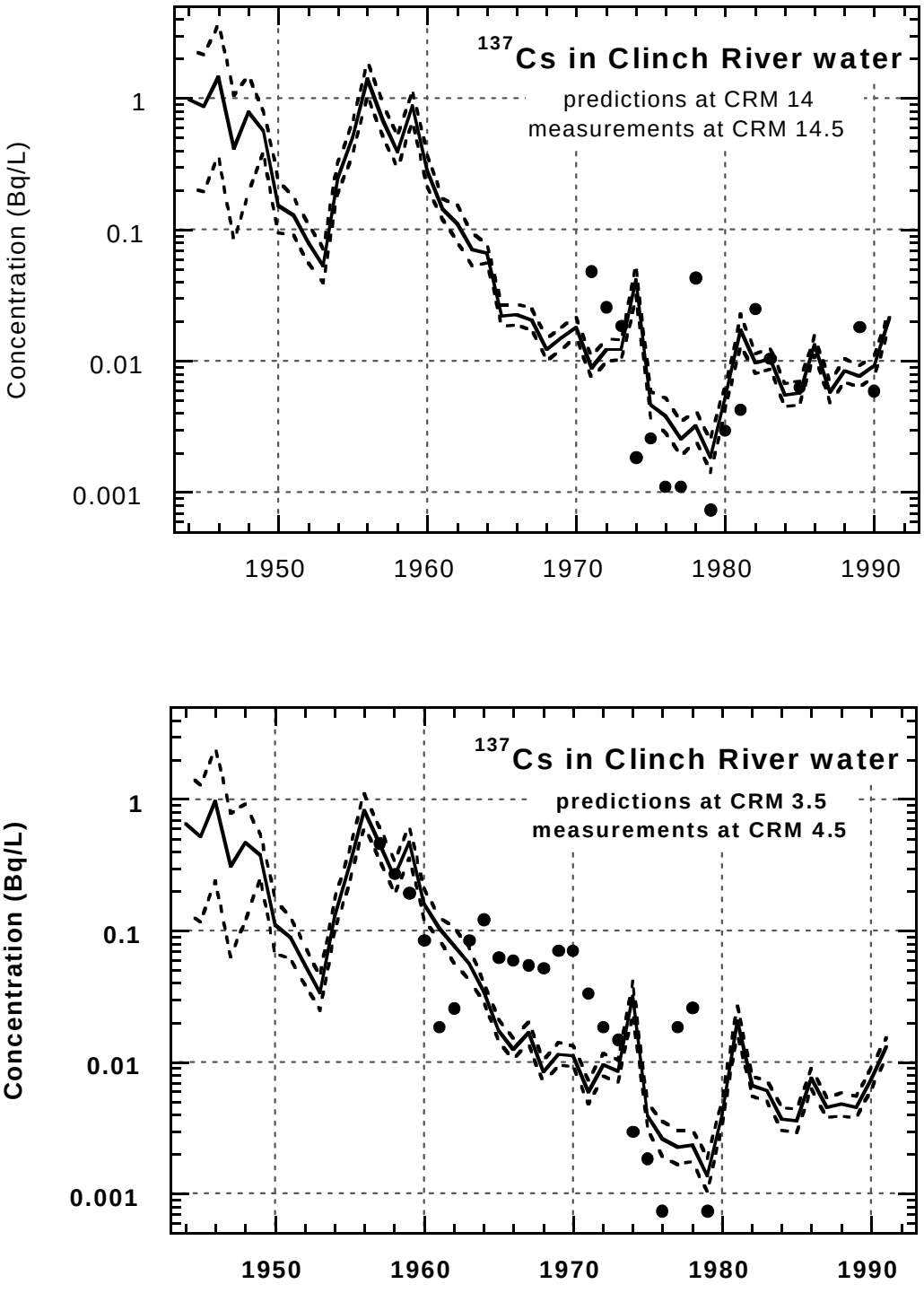

Figure 6.2 Comparison of predicted annual average concentrations of ^{137}Cs in water with measured annual average concentrations. Comparisons are shown for predictions at CRM 14 with measurements at CRM 14.5 (top) and for predictions at CRM 3.5 with measurements at CRM 4.5 (bottom). Solid lines indicate the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on uncertainty in release estimates. Dark circles indicate the measured values.

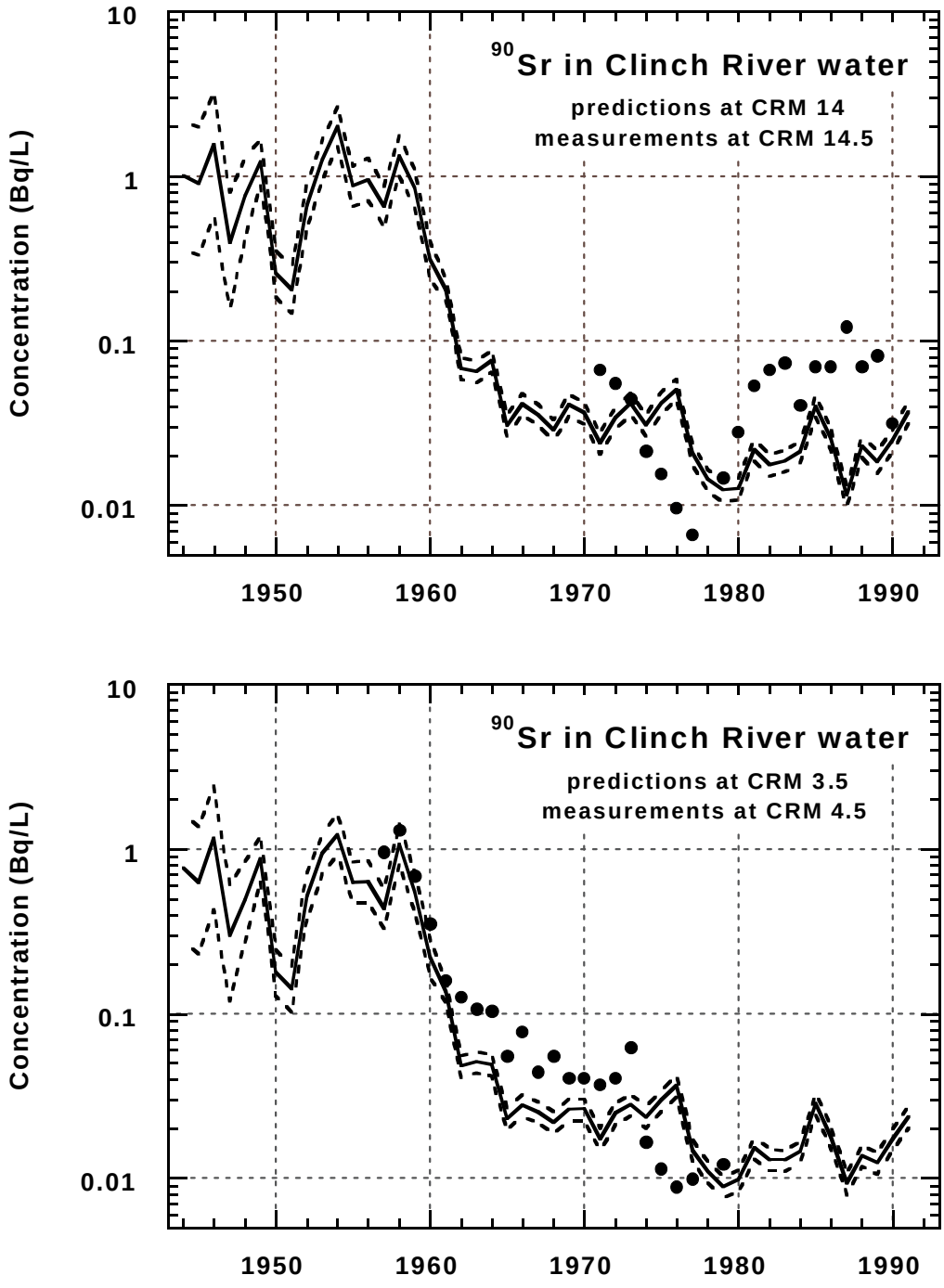

Figure 6.3 Comparison of predicted annual average concentrations of ^{90}Sr in water with measured annual average concentrations. Comparisons are shown for predictions at CRM 14 with measurements at CRM 14.5 (top) and for predictions at CRM 3.5 with measurements at CRM 4.5 (bottom). Solid lines indicate the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on uncertainty in release estimates. Dark circles indicate the measured values.

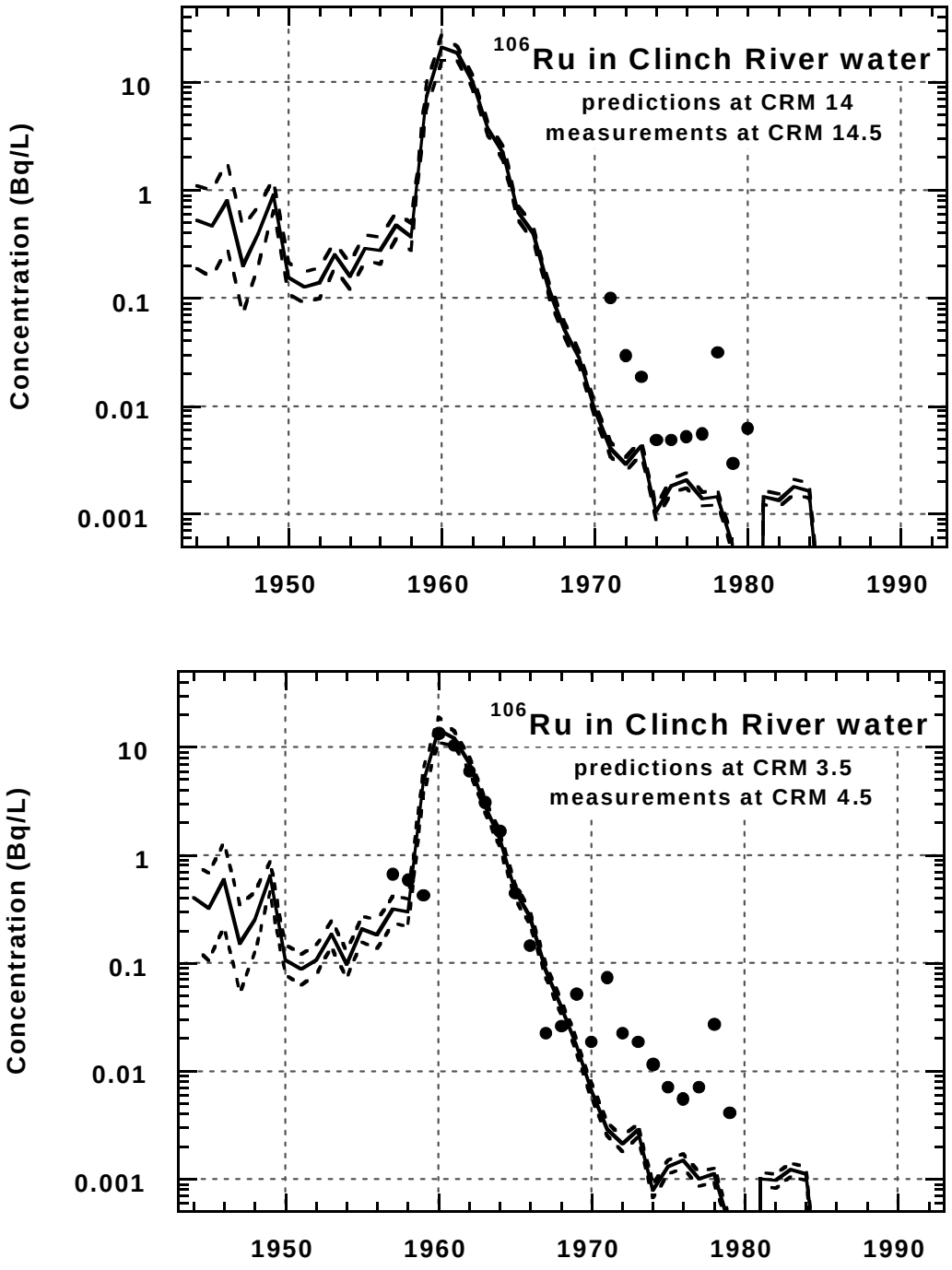

Figure 6.4 Comparison of predicted annual average concentrations of ^{106}Ru in water with measured annual average concentrations. Comparisons are shown for predictions at CRM 14 with measurements at CRM 14.5 (top) and for predictions at CRM 3.5 with measurements at CRM 4.5 (bottom). Solid lines indicate the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on uncertainty in release estimates. Dark circles indicate the measured values.

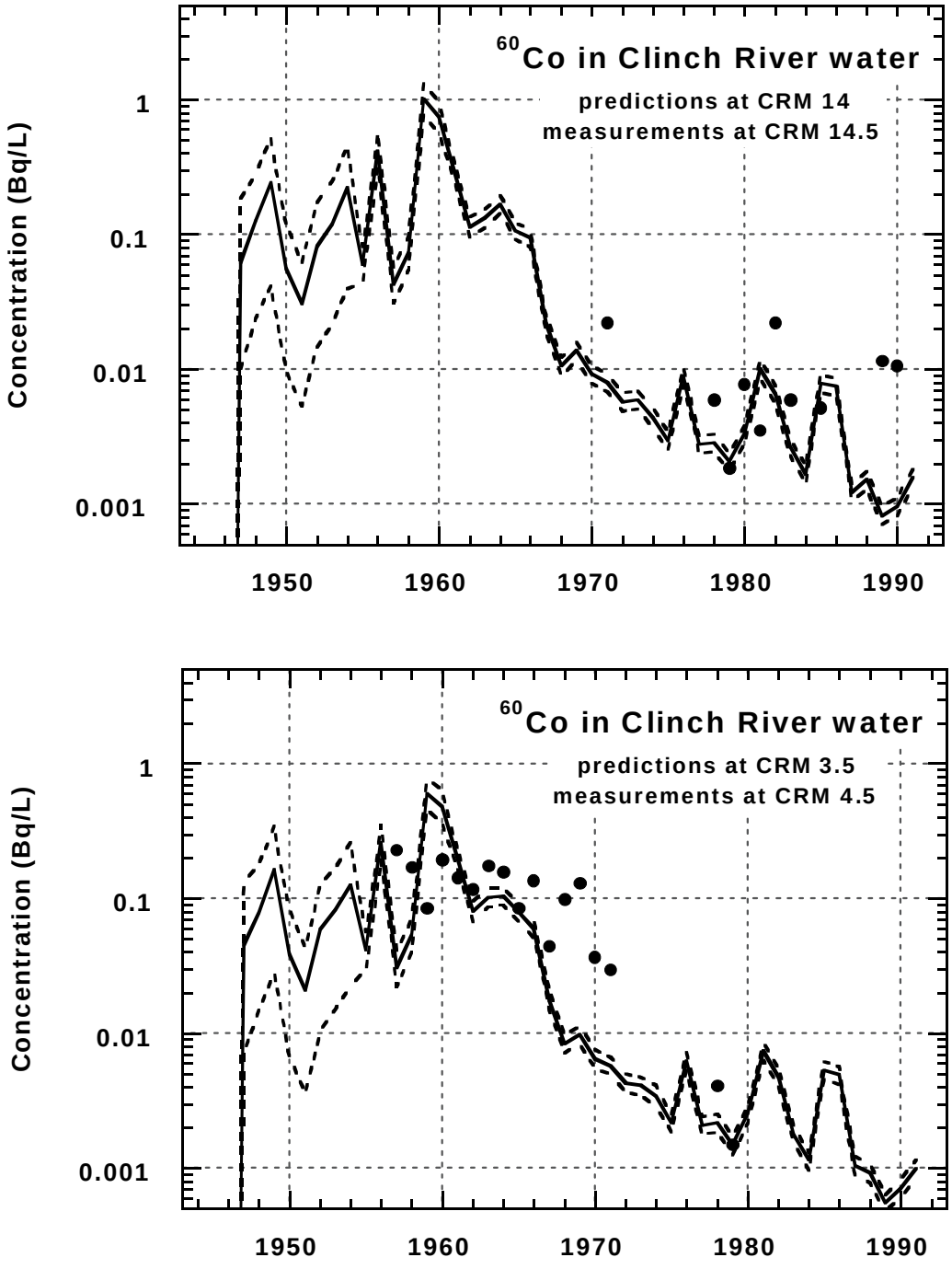

Figure 6.5 Comparison of predicted annual average concentrations of ^{60}Co in water with measured annual average concentrations. Comparisons are shown for predictions at CRM 14 with measurements at CRM 14.5 (top) and for predictions at CRM 3.5 with measurements at CRM 4.5 (bottom). Solid lines indicate the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on uncertainty in release estimates. Dark circles indicate the measured values.

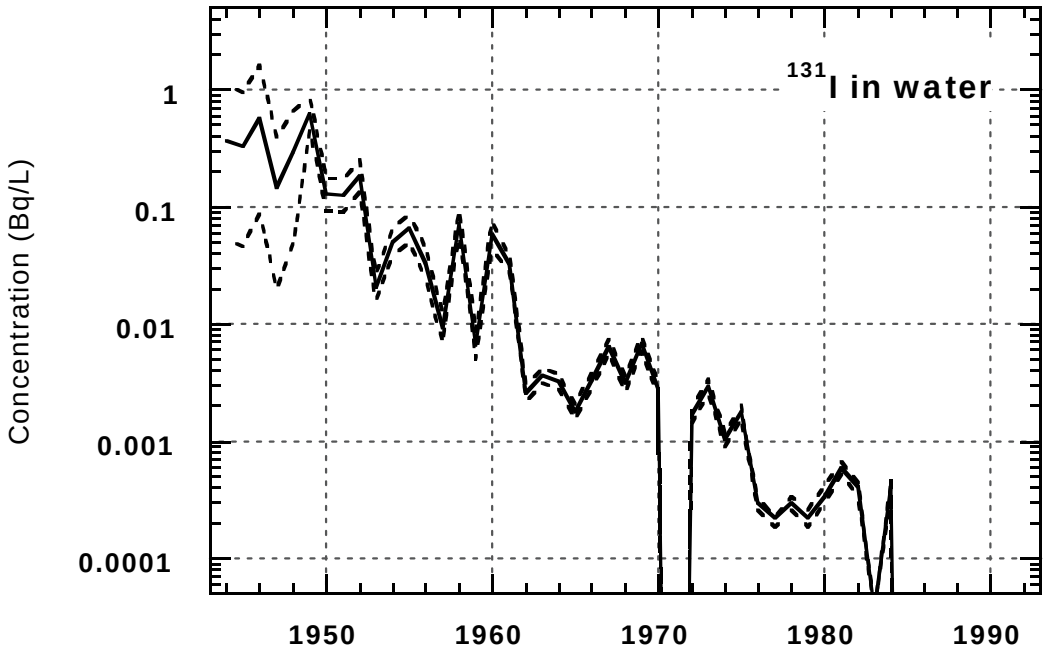

Figure 6.6 Example of predicted annual average concentrations of ^{131}I in water, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates and the K_d values.

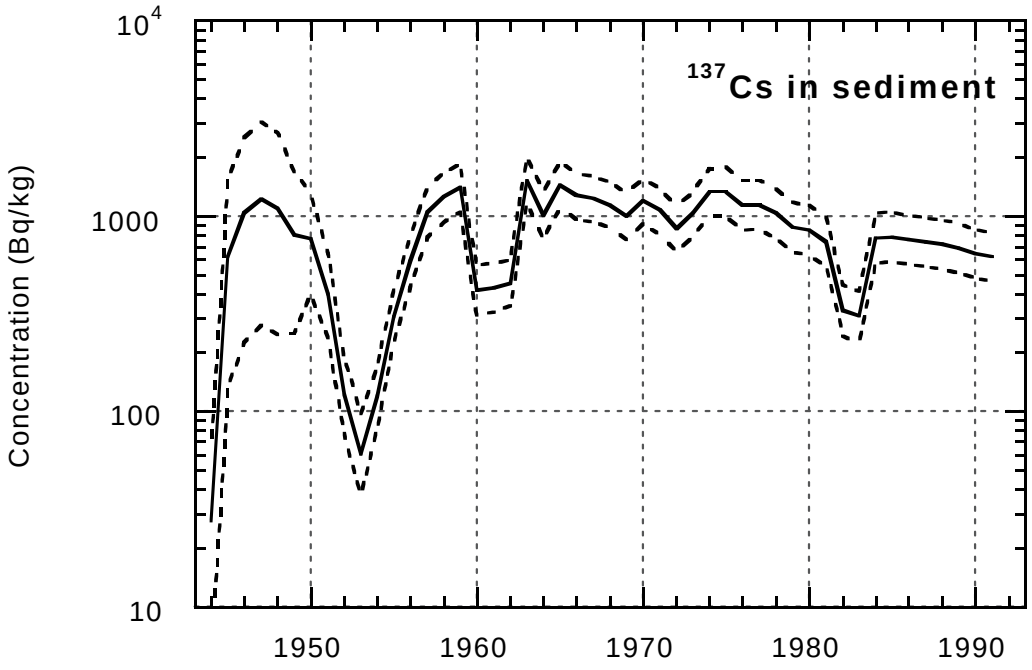

Figure 6.7 Example of predicted annual average concentrations of ^{137}Cs in shoreline sediment, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates.

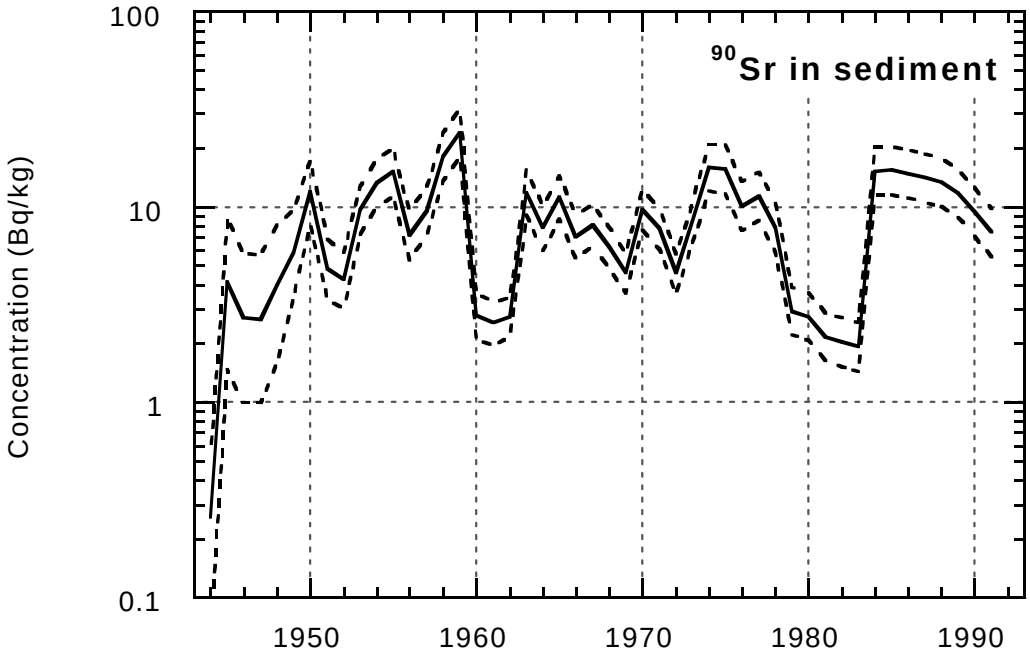

Figure 6.8 Example of predicted annual average concentrations of ^{90}Sr in shoreline sediment, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates.

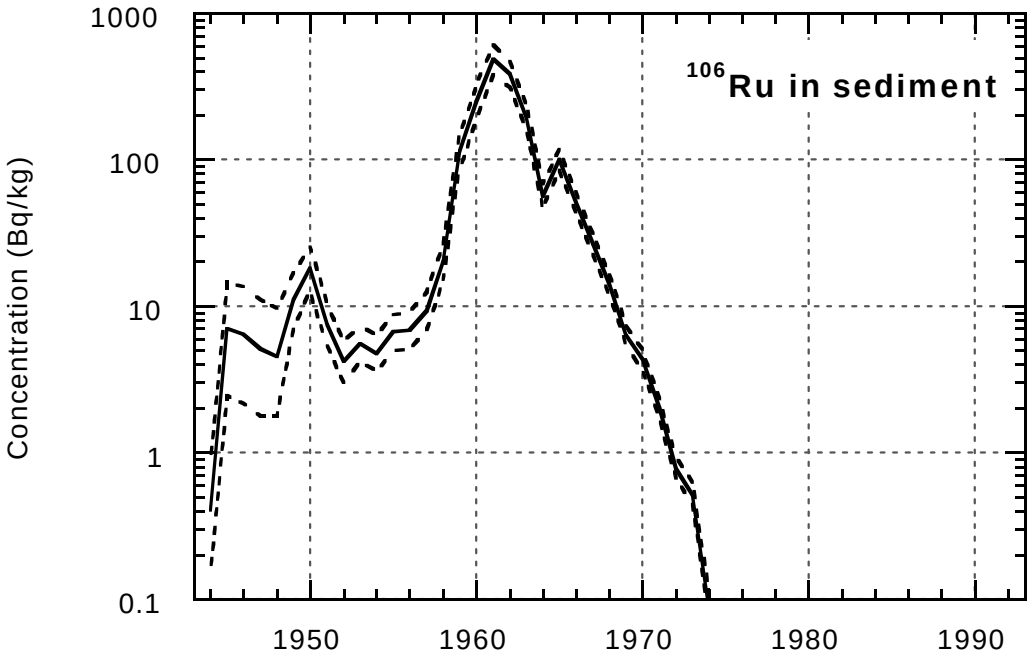

Figure 6.9 Example of predicted annual average concentrations of ^{106}Ru in shoreline sediment, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates.

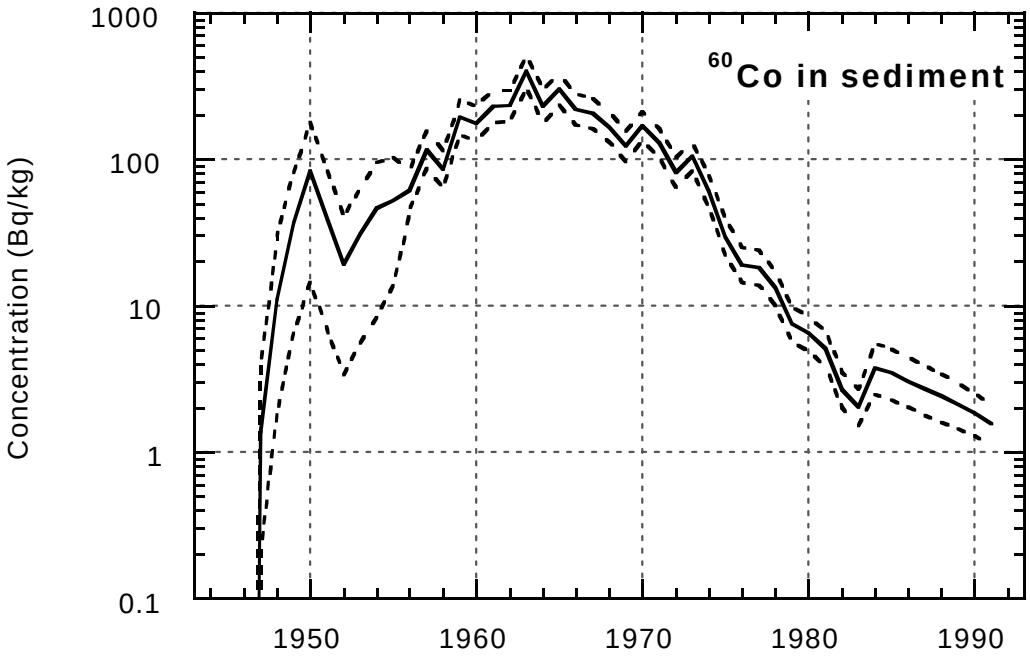

Figure 6.10 Example of predicted annual average concentrations of ^{60}Co in shoreline sediment, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates.

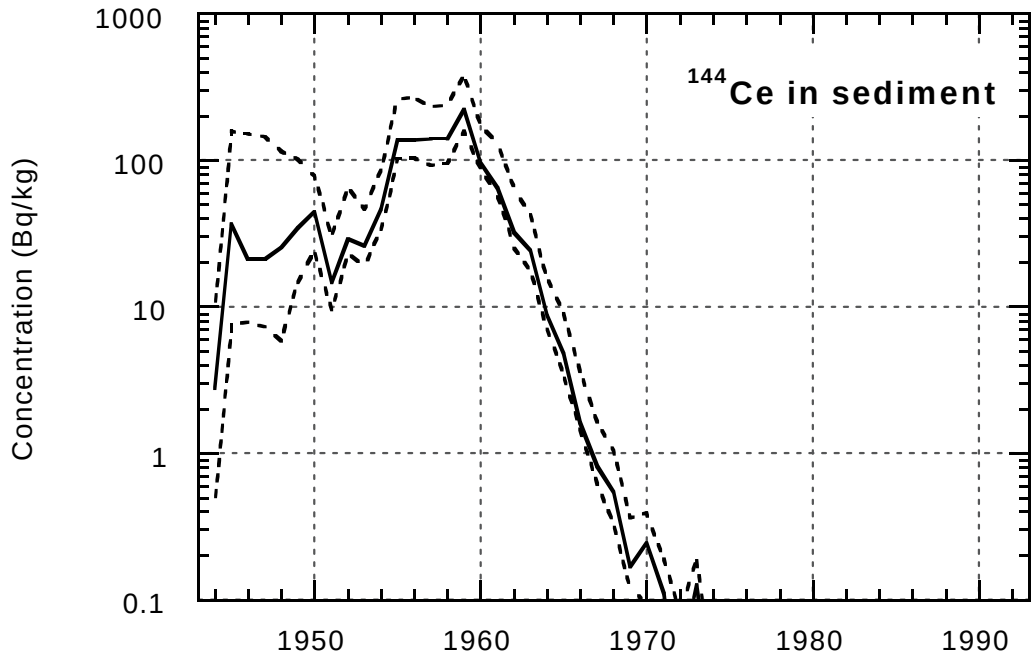

Figure 6.11 Example of predicted annual average concentrations of ^{144}Ce in shoreline sediment, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates and the K_d values.

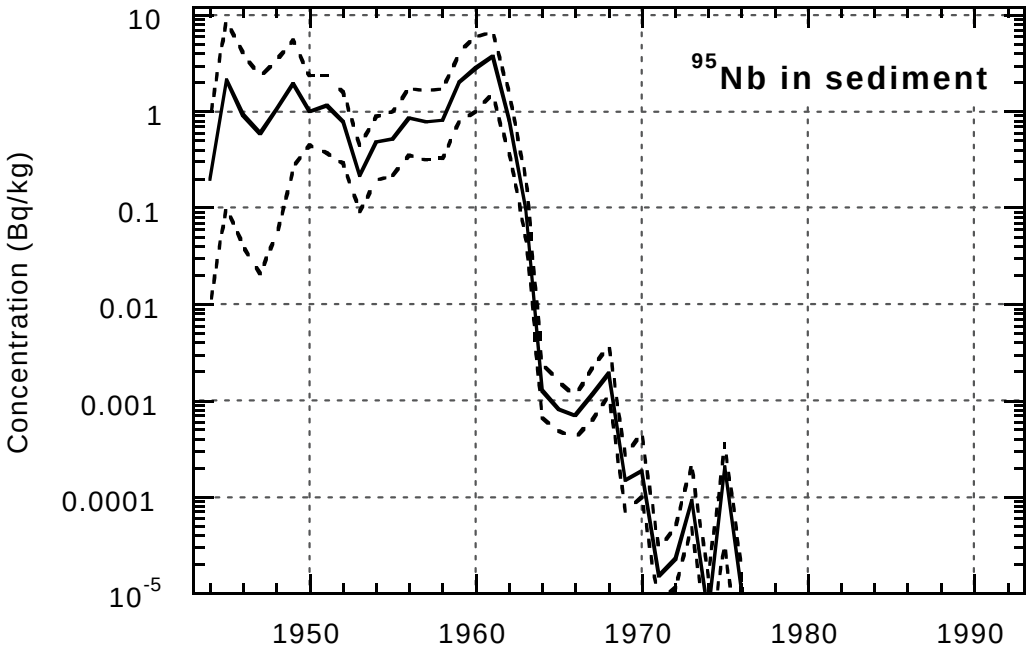

Figure 6.12 Example of predicted annual average concentrations of ^{95}Nb in shoreline sediment, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates and the K_d values.

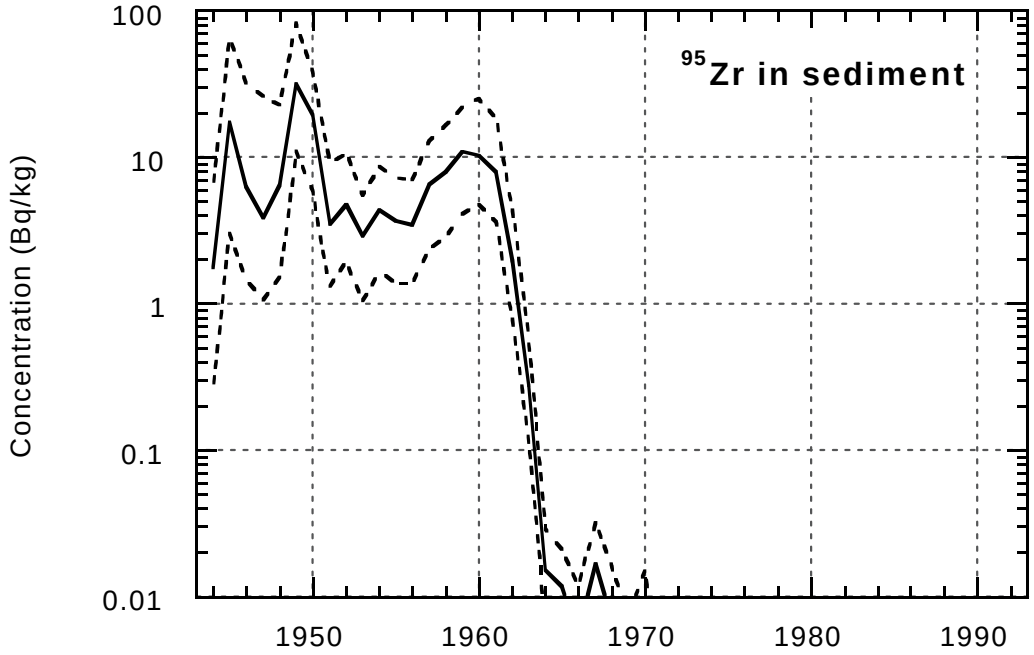

Figure 6.13 Example of predicted annual average concentrations of ^{95}Zr in shoreline sediment, shown for CRM 14. The solid line indicates the central values of the predictions; dashed lines indicate predicted 95% confidence bounds based only on the uncertainty in the release estimates and the K_d values.

An additional explanation for discrepancies between measured and modeled concentrations is the presence in the Clinch River of radionuclides from sources other than ORNL and White Oak Creek. Radionuclides from other sources would have been included in measurements but would not have been accounted for by the model. Three non-ORNL sources of radionuclides in the Clinch River are known: fallout from global atmospheric testing of nuclear weapons (primarily ^{90}Sr and ^{137}Cs); releases of radionuclides (primarily ^{60}Co) from the American Nuclear Corporation, located on a Clinch River tributary about 30 river miles upstream from White Oak Creek; and contamination from the K-25 or Y-12 sites transported via Poplar Creek. Poplar Creek is not expected to have contributed any significant amount of the radionuclides considered in this study.

The contribution of fallout is not a problem for most years, because the measured concentrations of ^{137}Cs and ^{90}Sr reported at locations upstream from White Oak Creek are significantly lower in most years than those measured at CRM 14.5 and CRM 4.5 for (Tables 6.1 and 6.10). However, Cowser and Snyder (1966) report that weapons testing fallout contributed about 45% of the ^{90}Sr and 20% of the ^{137}Cs found in the Clinch River during 1962 and 1963. The largest contributions of fallout occurred after 1960, while at this same time the releases from White Oak Dam were low compared to the pre-1960 releases.

The American Nuclear Corporation (ANC) used ^{60}Co as a radiation source for medical instruments between 1962 and 1970 (ORNL, 1992; Levine et al., 1994). ANC was located on Braden Branch, a small stream that enters the Clinch River at CRM 50.5. Although elevated levels of radioactivity were detected in the Clinch in the vicinity of Braden Branch, this source probably did not contribute significantly to ^{60}Co levels in the Clinch River below White Oak Creek (CRM 20.8 to CRM 0).

In general, the model appeared to give more accurate predictions for all radionuclides for the years prior to 1963 (1957-1962) than for the years after 1963. A logical explanation for this observation is that the model does not completely account for the backflow and scouring of sediments that occurred after Melton Hill Dam became operational in 1963. However, data from 1960-1963 may also reflect disturbances of the river due to construction of the dam, and data from 1957-1959 do not reflect a standard calender year.

Based on current knowledge, an additional uncertainty factor of 2 was applied to the water concentrations predicted by the HEC-6-R model. The confidence intervals on the HEC-6-R output reflects only the uncertainty in the source term and (for some radionuclides) the K_d values. The additional uncertainty factor accounts for discrepancies in the time scales of the model inputs (yearly releases and daily flows) and for the uncertainties in modeling localized scouring and deposition events. The uncertainty factor is expressed as a log-uniform distribution between 0.5 and 2 (midpoint 1). For years when the uncertainty in the source term is large, or when the range of K_d values is large, those uncertainties will continue to dominate. However, the uncertainty factor of 2 will increase the subjective confidence intervals for situations with low uncertainties on the release estimates and on K_d values, reflecting the increased importance of other factors (e.g., localized scouring events) on the uncertainty about the model results.

Table 6.10 Measured annual average radionuclide concentrations in the Clinch River at or above Melton Hill Dam (Bq L^{-1}).

	^{137}Cs	^{90}Sr	^{106}Ru	^{60}Co
1959[a]	-[b]	0.04	-	-
1960	-	0.03	-	-
1961	0.02	0.03	0.2	0.01
1962	0.007	0.06	0.3	
1963	0.007	0.05	0.3	0.004
1964	0.01	0.04	0.1	-
1965	0.01	0.02	0.09	-
1966	0.01	0.03	0.02	-
1967	0.004	0.01	0.01	-
1968	-	0.01	0.02	0.2
1969	0.03	0.03	0.03	0.1
1970	0.01	0.02	0.01	0.01
1971	0.004	0.02	0.04	0.01
1972	0.004	0.02	0.02	-
1973	0.01	0.02	0.01	-
1974	0.001	0.003	0.004	-
1975	0.0007	0.003	0.003	-
1976	0.0007	0.003	0.005	-
1977	0	0.006	0.004	-
1978	0.009	0.003	0.02	0.003
1979	0.0004	0.0004	0.002	0.0004
1980	0.003	0.004	0.006	0.004
1981	0.003	0.05	0.001	0.008
1982	0.008	0.05	-	0.006
1983	0.003	0.035	-	0.003
1984	<0.01	0.01	-	<0.01
1985	0.041	0.029	-	<0.01
1986	<0.01	0.02	-	<0.01
1987	<0.21	0.44	-	<0.19
1988	<0.12	0.13	-	<0.56
1989	-0.20	0.063	-	0.11
1990	0.22	0.081	-	0.13
1991	0.085	-	-	-0.14

[a] Fourth quarter of 1959
[b] not reported

The central values and 95% confidence intervals of the model predictions (by radionuclide and year; Appendix 6A) were approximated by lognormal distributions, based on the 2.5th and 50th percentiles of the model predictions. These distributions represent the uncertainty in the source term and K_d values. The lognormal distributions thus expressed were multiplied by the additional uncertainty factor (expressed as a log-uniform distribution from 0.5 to 2) to obtain a final distribution on the predicted water concentrations.

The final subjective distributions represent all sources of uncertainty about the predicted water concentrations. Approximation of the model predictions by lognormal distributions decreased the calculational complexity of the assessment. However, for a few situations where the model predictions were asymmetric about the median (primarily the years 1944-1948 for all radionuclides), this results in a slight inflation of the upper bound on the water concentrations for those years. This is not expected to have a significant effect on the estimates of dose and risk.

As discussed in Section 6.3, modeled water concentrations were used in this analysis only when measurements were unavailable or inaccurate. These periods included 1957-1959 for all radionuclides (when measurements were not reported on a calendar year basis) and years when "less-than" values were reported (Table 6.1), as well as years prior to 1957. (The 1957-1959 measurements are shown in Figures 6.2-6.5, even though they were not used in the analysis.) Appendix 6B contains tables of the final water concentrations that were used in the analysis. For ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co, these are based on measurements to the extent possible (Section 6.3) and otherwise on modeled values. Concentrations of ^{131}I in water are based entirely on model results. The confidence limits given in Appendix 6B include the uncertainty factors described above and in Section 6.3.

6.13 Sources of Uncertainty in Modeled Sediment Concentrations

In addition to the sources of uncertainty described in Section 6.2 for predicted water concentrations, which could also affect predicted sediment concentrations, additional uncertainty exists concerning the concentrations of radionuclides in sediment that a person would be exposed to at any given time due to varying positions of the water level and varying radionuclide concentrations in the sediments up or down the shoreline (perpendicular to the river). Therefore, the modeled concentrations at all locations, for which the distributions represent only the uncertainty in the source term and (for some radionuclides) the K_d values, were multiplied by an additional uncertainty factor (expressed as a log-uniform distribution between 0.33 and 3) to ensure that the true radionuclide concentrations in shoreline sediments that people might have actually encountered were properly encompassed. This uncertainty factor accounts for the uncertainties in modeling localized situations and will affect the width of the resulting subjective confidence interval primarily when the uncertainties in both source term estimates and K_d values are low. The central values and 95% confidence intervals of the model predictions (by radionuclide and year; Appendix 6A) were approximated by lognormal distributions, based on the 2.5th and 50th percentiles of the model predictions; these lognormal distributions (which represent only the uncertainty in the source term and K_d values) were then multiplied by the additional uncertainty factor to obtain the final sediment concentrations that were used in the analysis (Appendix 6C). The final subjective distributions represent all sources of uncertainty about the predicted sediment concentrations. Approximation of the model predictions by lognormal distributions decreased the calculational complexity of the assessment. However, for a few situations where the model predictions were asymmetric about the median, this results in an inflation (1944-1948 for all radionuclides) or reduction (1982-1991 for ^{144}Ce) of the upper bound on the sediment concentrations for those years. Sample calculations indicate that this inflation or reduction of the upper

bound on the predicted sediment concentrations will not have a significant effect on the estimates of dose and risk.

Only one set of measured radionuclide concentrations in shoreline sediment was available for comparisons with the concentrations predicted by HEC-6-R. Between 1991 and 1994, approximately 300 samples of near-shore surface sediment were collected along the Clinch River and analyzed for ^{137}Cs and ^{60}Co (Levine et al., 1994). Samples below the detection limit were set equal to the detection limit for calculation of the mean, causing a bias toward higher concentrations than actually exist; this is important only for the ^{60}Co concentrations, especially downstream of the Clinch River. In general, concentrations were slightly higher for the lower Clinch River (CRM 0-12.0) than for the upper Clinch River (CRM 12.1-23.0), but noticeably lower in the Tennessee River (Watts Bar Reservoir) below the confluence of the Clinch than in the Clinch (Table 6.11; the high standard deviations occur because the values are lognormally distributed). The slightly higher concentrations in the lower Clinch River than in the upper Clinch are attributed to greater deposition of fine particles downstream of the Poplar Creek confluence (Levine et al., 1994).

The predicted mean concentrations of ^{137}Cs and ^{60}Co in shoreline sediment in 1991, with the uncertainty factor described above, were compared with the measurements by location along the Clinch River (Fig. 6.14; the observed mean, minimum, and maximum values are indicated). The graphs indicate that the predicted mean concentrations (indicated by a 95% subjective confidence interval) are of the same general magnitude as the observations for each radionuclide, supporting the general validity of the model predictions. It should be pointed out that, although the measurements were taken between 1991 and 1994, the predicted concentrations in 1991 and the several preceding years are fairly stable, except for the effect of radioactive decay (most noticeable for ^{60}Co; Appendix 6C). The comparison therefore is not an entirely unreasonable one. It should also be mentioned that the measured concentrations are primarily but not entirely for near-shore surface sediment, both above and below the waterline; some samples were taken in the river channel rather than near the shore. In addition, the measurements do not represent any particular depth of the surface sediment, but rather a grab sample of a depth that depended on how soft the bottom was at that point; some samples may represent the top 30 cm or so of the sediment layer.

The measured sediment data do show a decrease in radionuclide concentration with distance up or down the bank (perpendicular to the river). This observation, together with the fact that the water level is generally lower in the winter, suggests that a person on the shoreline is exposed to a low concentration in the summer, when the water level is high, and a higher concentration in the winter, when the water level is low. Although calculations for external exposure to shoreline sediments (Section 10) included a term for amount of time exposed during high-water or low-water seasons, the modeled concentrations in sediment include only yearly averages.

Table 6.11 Summary of measured radionuclide concentrations in shoreline sediment in 1991-1994 (Levine et al., 1994).

Radionuclide and location	Number of samples[a]	Mean (Std. Dev.) Bq kg^{-1}	Range Bq kg^{-1}
Cs-137			
CRM 12.1-23.0 (Melton Hill Dam to Poplar Creek)[b]	166 (0)	93.5 (184)	2.11-1400
CRM 0-12.0 (Poplar Creek to Tennessee River)	156 (0)	107 (106)	6.75-807
TRM 530-567.5 (Clinch River to Watts Bar Dam)	496 (2)	19.2 (30.6)	0.522-382
Co-60			
CRM 12.1-23.0 (Melton Hill Dam to Poplar Creek)[b]	166 (37)	3.25 (2.64)	0.540-19.4
CRM 0-12.0 (Poplar Creek to Tennessee River)	156 (7)	4.87 (3.62)	0.810-19.1
TRM 530-567.5 (Clinch River to Watts Bar Dam)	496 (323)	1.28 (1.18)	0.089-10.0

[a] Number in parentheses is the number of samples below detection limits.

[b] Approximately 85% of these samples were taken below the mouth of White Oak Creek (between CRM 12.1 and CRM 20.8).

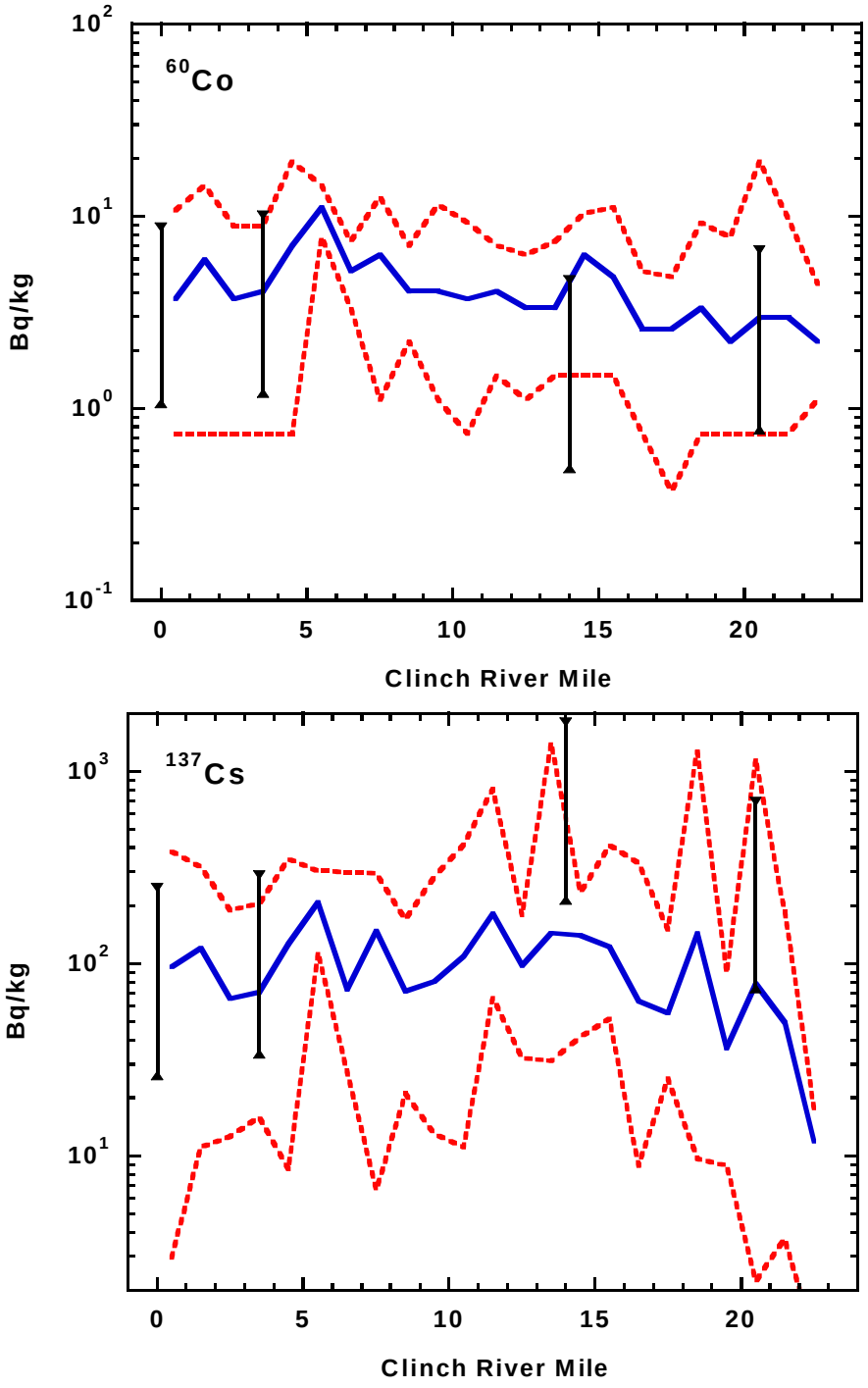

Figure 6.14 Comparison of predicted mean concentrations of ^{137}Cs (top) and ^{60}Co (bottom) in Clinch River shoreline sediments in 1991 with measurements of near-shore surface sediment taken in the Clinch River between 1991 and 1994. The thick solid line indicates the observed mean value from samples taken at the indicated locations; the dashed lines represent the minimum and maximum observed values (except for ^{137}Cs at CRM 6, for which only one measurement was reported). The vertical lines represent the 95% subjective confidence interval on the predicted mean concentration for the indicated locations.

6.14 Summary

The evaluation of potential health risks from past releases of radionuclides from White Oak Creek was based on estimated annual average concentrations of radionuclides in Clinch River water and shoreline sediments from 1944 to 1991. Environmental measurements of ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co in Clinch River water were used as available for 1960-1990 (Section 6.3); estimates of radionuclide concentrations in Clinch River water obtained with the HEC-6-R model were used for the remaining radionuclides and years (Section 6.12). Concentrations of radionuclides in shoreline sediments were estimated with the HEC-6-R model for all radionuclides and all years (Section 6.13). The concentrations of ^{60}Co, ^{90}Sr, ^{106}Ru, and ^{137}Cs used in the analysis, after all uncertainties and adjustment factors were propagated, are given in Appendix 6B (water) and Appendix 6C (sediment) for the Clinch River locations of interest. While in many cases the modeled and measured values were comparable, the concentrations based on measurements generally reflect a higher degree of confidence (lower uncertainty) than do the modeled concentrations. It should be noted, however, that the measurements include radionuclide contamination from all sources; in particular, measurements of ^{137}Cs and ^{90}Sr reflect the contribution from global weapons testing fallout as well as from any scouring events that were not accounted for by the model.

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7.0 ASSESSMENT OF RELEVANT EXPOSURE PATHWAYS

Sections 7.1 to 7.7 describe the relevant exposure pathways as determined from demographic information, an initial screening analysis, and the risk-based decision criterion established by the Oak Ridge Health Advisory Steering Panel (ORHASP). For this study, the lower Clinch River was divided into five segments or reaches: Clinch River Mile (CRM) 21 to CRM 17 (Jones Island), CRM 17 to CRM 14 (Grassy Creek), CRM 14 to CRM 5 (K-25), CRM 5 to CRM 2 (Kingston Steam Plant), and CRM 2 to CRM 0 (City of Kingston). Topography and land use for each reach are described, followed by a description of the potential exposure pathways for that reach. The target individuals who were potentially exposed to the identified pathways and the parameters associated with these pathways are described in Section 7.8. The estimated number of individuals exposed via these pathways is described in Section 7.9. A summary of the main demographic findings for the lower Clinch River area is provided in Section 7.10.

7.1 Identification of Locations of Interest and Potential Exposure Pathways

Several locations have received attention in the determination of the land use along the Clinch River, including Jones Island, Grassy Creek, K-25, Kingston Steam Plant, and the City of Kingston. These areas of interest are located within Roane County and have been and are still primarily agricultural and residential areas (USDOE, 1996). A map of the Oak Ridge Reservation (ORR) with locations of interest for the dose reconstruction for radionuclides from White Oak Creek is presented in Figure 7.1.

The Tennessee Valley Authority (TVA) purchased all the shoreline along the south side of the Clinch River from approximately Melton Hill Dam to Watts Bar Dam prior to the filling of Watts Bar Reservoir, which occurred in 1942 (TVA, 1987). This TVA land, which was purchased in 1941, is indicated as flood easement property on Roane County Courthouse records and maps (Brown, 1996). Even though TVA owned the shoreline, property adjacent to the south bank of the Clinch River was used for farming prior to 1949 (Waller, personal communication with C. Lewis, 1996; Brown, 1996; Prichard, personal communication with C. Lewis, 1997). The residents of these farms and their farm animals had access to the river, since there was no fence separating the TVA land from the private land (Prichard, personal communication with C. Lewis, 1997).

The land located on the northern side of the river is owned by the Atomic Energy Commission (now known as the U.S. Department of Energy, USDOE) and TVA from approximately Clinch River Mile (CRM) 34.0 to CRM 10.0 (USGS, 1989; 1990). Fifty-nine thousand acres of land, primarily on the north side of the Clinch River, was purchased early in 1942 by the Army Corps of Engineers as part of a federal reserve, where development of the atomic bomb took place (Krause, 1992; Jackson, 1981). About 3000 residents living on this designated land received court orders to vacate, within weeks, the land their families had owned and farmed for generations (Krause, 1992; Jackson, 1981).

Several islands (Jones Island, Grubb Island, and Brashear Island) are located in the Clinch River. The Jones family as part of a land grant originally obtained Jones Island (Waller, personal communication with C. Lewis, 1996). TVA purchased approximately 26 acres of the island as part of the Watts Bar flow easement in 1941 (Title Search, 1996). The Jones family owned the remaining property and their heirs until it was sold to individuals in 1944 (Title Abstract, 1962). Individual owners retained possession of the island until the Melton Hill Project was initiated in 1963. TVA then purchased the remaining acreage (approximately 15 acres) on the island in 1963 (Warranty Deed, 1963).

In the initial screening analysis (see Section 3), all possible potential exposure pathways were examined. These pathways included fish ingestion, swimming (external exposure and inadvertent consumption of river water), drinking water, external exposure from shoreline sediment, and external exposure from dredged sediment. Three agricultural pathways were also considered in the screening analysis: ingestion of produce or other crops grown on land contaminated by irrigation or soil enrichment with dredged sediment; ingestion of milk from dairy cattle; and ingestion of beef from beef cattle. Both dairy and beef cattle may have been exposed via irrigation of pasture or crops or via drinking water. Based on a conservative screening analysis (see Section 3), two of the potential exposure pathways were assigned low priority for further study (i.e., swimming and ingestion of produce contaminated via irrigation).

Demographic information was obtained through personal interviews and telephone interviews with Clinch River community members living in the area from the early 1900s until the present. This demographic information was then used to further evaluate the likelihood of the potential exposure pathways. Based on this information, the potential exposure pathways considered in this assessment for the area near and adjacent to the lower Clinch River included fish ingestion, drinking water, ingestion of milk and meat, and external exposure from shoreline sediment.

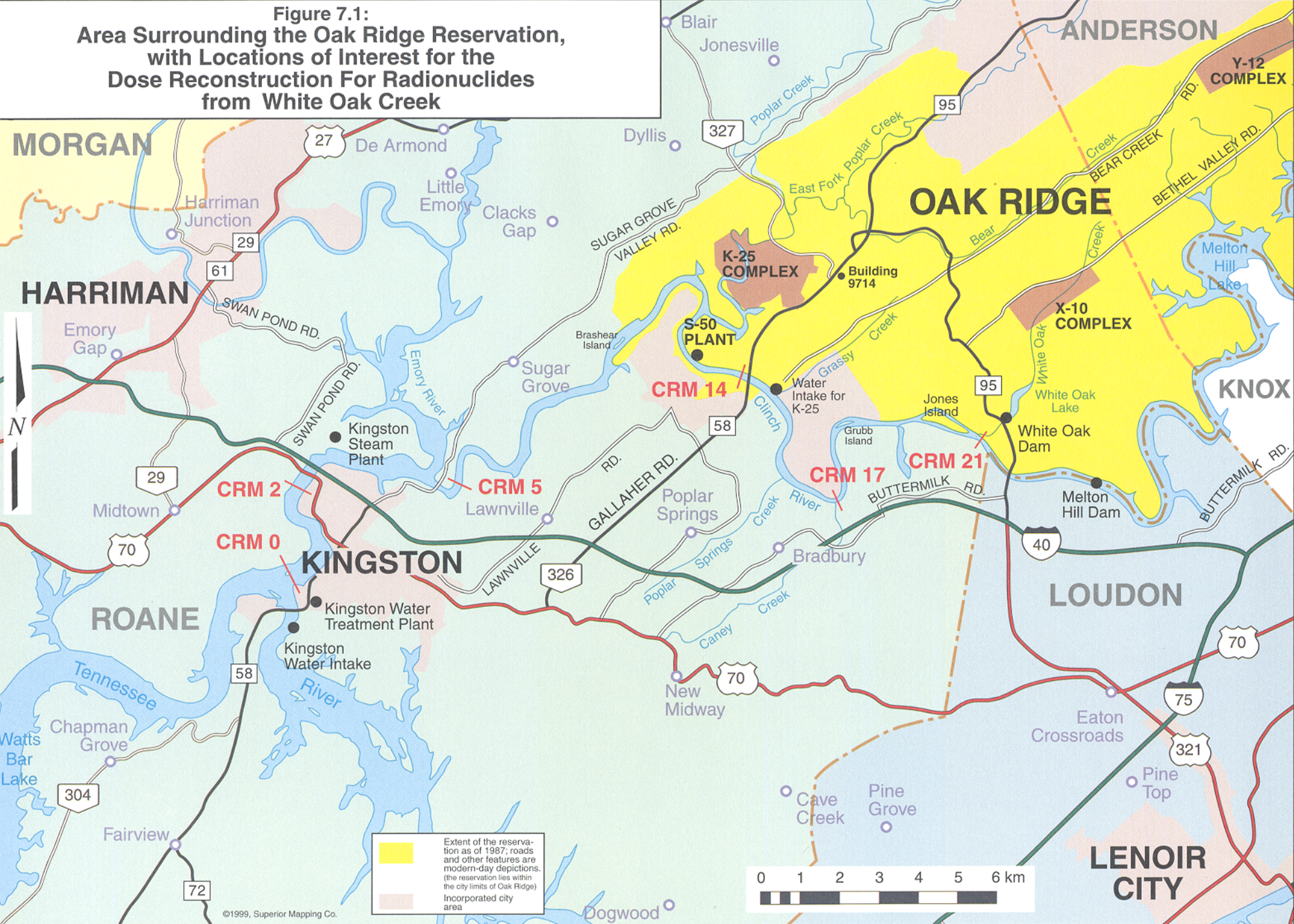
7.2 Jones Island Area - CRM 21.0 to CRM 17.0

The Jones Island area is considered to include the island itself and the land adjacent to the banks of the Clinch River from CRM 21.0 to CRM 17.0 (Figure 7.1). The topography in this area consists of rolling land with gradual slopes. The primary activities conducted along this section of the river during the 1940s and 1950s were agricultural.

7.2.1 Description of Land Use in the Jones Island Area

Jones Island (also called Blue Springs Island), a small land formation located in the Clinch River at approximately CRM 20.0, is an area of primary concern because of its proximity to the mouth of White Oak Creek. This island has been used in the past as beef cattle grazing land, for growing silage crops, and for archeological digs. The island was used by the Jones family for grazing beef cattle and for producing silage, primarily corn and hay. The land was in the possession of the Jones family heirs for approximately 45 years, before it was sold to individuals in 1944 (Title Abstract, 1962). From 1944 to 1962, various individual owners owned the island (Title Abstract, 1962).

Figure 7.1:
Area Surrounding the Oak Ridge Reservation, with Locations of Interest for the Dose Reconstruction For Radionuclides from White Oak Creek



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During this time period, the island was covered predominantly in a sod of clover and grass mixtures and was used for hay and pasture (Field Appraisal, 1962). A small stock barn and a partially completed structure were erected on Jones Island sometime during this period (Field Appraisal, 1962). The last individual owner of the island indicated that tobacco was grown on the island for a one-year period, but the crop was not productive (Huber, personal communication with C. DaMassa, 1996). Beef cattle were also grazed on the island during the spring and summer months. In addition, an archeological dig was conducted on the island in 1962 as part of the planning process for the Melton Hill Dam project (Frankenburg, personal communication with W. Reed, 1996).

Since construction of the dam would cause flooding in areas along the river in the Jones Island area, TVA purchased the island as part of the Melton Hill Dam project in January, 1963 (Warranty Deed, 1963). The individual owners had until December 1963 to remove all possessions from the island; however, TVA could use the land for the purpose of disposing dredged sediment any time after the purchase (Contract for Purchase and Sale of Land, 1962). The island was altered by dredging that occurred in 1963 (TVA, 1966) to improve the river channel for barge traffic. According to a TVA report, a suction dredge was used to remove 24,100 cubic yards of silt, sand, and gravel in the Jones Island area (TVA, 1966). Once TVA purchased the land, the land was no longer used for farming activities and was not leased to any individuals (Robinson, personal communication with C. Lewis, 1996).

The land areas on both sides of the river were also altered when TVA began construction of Melton Hill Dam (CRM 23.1) in 1963. The land lying above the dam on the north side of the Clinch River was owned by the Atomic Energy Commission (AEC) and was flooded by the Melton Hill Reservoir (TVA, 1966). The remaining (AEC) farmland below the dam was of marginal value; therefore, the AEC abandoned the area (TVA, 1966). Since the land was abandoned, no farm access roads were constructed, and no additional farming activities occurred.

The southern bank of the Clinch River in the Jones Island area was also affected by the release of radioactive material from White Oak Creek after the dam went into operation in 1963. The Jones Island area on the south bank of the Clinch River (the area directly across from Jones Island) was not affected by releases from White Oak Creek until 1963, because the water was constantly moving downstream, not pooling and stagnating upstream and at the mouth of the creek. Prior to 1963, the releases from White Oak Creek remained unmixed until after the water had passed Jones Island (Morton, 1966). The unmixed releases from White Oak Creek traveled primarily along the northern bank of the river. Complete mixing of the water did not occur until after it passed Jones Island, in approximately the Grassy Creek area (Morton, 1966). With the installation and operation of Melton Hill Dam, the water being released from White Oak Creek was traveling upstream (backflowing), rather than downstream, when the dam was not in operation. When water was released from the dam, the water from White Oak Creek was mixed with the released water in the vicinity of the dam. The mixed water would then flow past White Oak Creek and along both sides of Jones Island, thus affecting the southern bank of the Clinch River in this area.

The land on the Clinch River lying to the south of the island was used for farming as early as the 1900s (Waller, personal communication with C. Lewis, 1996). After the Civil War, the land found between approximately CRM 21 and CRM 19 was obtained by land grants and was owned by four large families (Waller, personal communication with C. Lewis, 1996). These families raised cattle and swine primarily, along with a few chickens and dairy cattle on each farm (Waller, personal communication with C. Lewis, 1996; Prichard, personal communication with C. Lewis, 1997).

Hereford beef cattle were the most popular breed in the area, with an average herd size of approximately 200 (Waller, personal communication with C. Lewis, 1996). The Herefords grazed on the pastureland but were fattened with the grain and hay produced on the farms. Since the cattle were primarily grazing away from the barns, water sources consisted of small creeks, ponds, and the Clinch River, if accessible. However, several of the large land grant farms did not have river frontage, so the Clinch River was not a viable water supply for the animals (Waller, personal communication with C. Lewis, 1996). The beef cattle were raised as a cash crop rather than for family consumption. AFarmers may have reserved for the family less meat and poultry than was wanted because of the high financial return for their sale@ (USDA, 1944). Once the cattle reached a suitable weight, they were sold at auction (Huber, personal communication with C. DaMassa, 1996; Waller, personal communication with C. Lewis, 1996).

The dairy cattle were primarily Jersey cattle, with each farm having 5 or 6 animals (Waller, personal communication with C. Lewis, 1996). The milk obtained in this area along the Clinch River was used for family consumption and for making butter and cheese (Waller, personal communication with C. Lewis, 1996). Any excess milk or milk products were sold to the Charles H. Bacon Dairy in Loudon, Tennessee (Waller, personal communication with C. Lewis, 1996).

Pork was the primary meat source for the residents along this section of the river. The swine were slaughtered each fall, and the meat smoked and maintained in a smokehouse until needed (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). Although swine were raised along the Clinch River, they did not consume river water. These animals were kept in pens located near the barns, which were generally close to the family dwelling. The swine were generally fed scraps, garden remnants, and milk that was not used by the family. The water that was consumed was provided by springs, creeks, or streams that ran through or adjacent to the pens (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). If these water sources were not available, well water was provided for these animals (Prichard, personal communication with C. Lewis, 1997).

In addition, chickens and goats were also raised along the Clinch River. The chickens were allowed to roam freely around the farm, but generally remained near the house or barn. The birds preferred these areas for feeding purposes: grain dropping out of the animal troughs and corn scattered on the ground for them by the farmers. Also, chickens do not require large amounts of water (Cole and Ronning, 1974), so they primarily obtained water from nearby creeks or streams. Chicken was used as a meat source that provided variety to the area farmers= diet (Waller, personal communication with C. Lewis, 1996).

Goats were kept on several farms in this area for land clearing purposes. These animals were used to remove unwanted growth from areas needed for agricultural purposes (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). One family in the Kingston area (outside the city limits and with no river access) raised and milked their goats (Prichard, personal communication with C. Lewis, 1997). The family used the goat milk and provided it to a few close neighbors (Prichard, personal communication with C. Lewis, 1997). Goat milk was not sold commercially.

Beef cattle were the only animals raised on farms in the lower Clinch River area that were used for human consumption and that had river access. Therefore, beef is used as a surrogate for all meat in this analysis. Although farmers consumed other types of meat (pork and chicken), the only type considered to be contaminated by radioactive effluents from the Oak Ridge Reservation to the Clinch River was beef.

Almost all food consumed by the families was grown on the farm. Each farm had a large vegetable garden that contained beans, beets, corn, and various root crops (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). The cash crops consisted of wheat, corn, and hay. Some of the grain crop and the hay were used as cattle feed. However, the community produced enough produce and grain to support a general store and gristmill. The Waller Family owned the gristmill, currently known as The Cross-Eyed Cricket. Local farmers had their wheat and corn ground into flour and meal, respectively (Lockwood, personal communication with W. Reed, 1996). The excess flour and meal were then sold at the Waller General Store to other local families and tenants (Waller, personal communication with C. Lewis, 1996). No milk, milk products, or meat was sold in this general store due to the lack of refrigeration (Waller, personal communication with C. Lewis, 1996). The tenants who lived on the larger farms in this area also grew tobacco as a cash crop, in addition to grains.

7.2.2 Potential Exposure Pathways in the Jones Island Area

The potential exposure pathways considered for this reach of the Clinch River include fish ingestion, external exposure from shoreline sediment, external exposure from dredged sediment, ingestion of meat and milk from cows, ingestion of produce contaminated via irrigation, and ingestion of produce contaminated via dredged sediment.

7.2.2.1 Fish Ingestion

Tenants (individuals who helped farmland owned by others in exchange for food and housing) who lived on the larger farms were the primary consumers of fish along this section of the Clinch River. Most of the larger landowners preferred pork for meals and did not consume fish on a regular basis (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). The tenants, however, used trotlines to obtain fish from the Clinch River on a regular basis. The children set the lines and then retrieved the fish later. Catfish was the most popular fish caught, but other species (for example, crappie and white bass) were also caught and consumed (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996; Prichard, personal communication with C. Lewis, 1997). Commercial fishing in this area was not mentioned in any interview and was unlikely due to

limited access for boats (Ebert, 1996). Since the consumption of fish was identified as an important pathway during the screening assessment and tenants living in this area used fish as a meat source, the fish ingestion pathway is considered in the risk assessment.

7.2.2.2 External Exposure from Shoreline Sediment

Exposure to shoreline sediments would occur primarily as a result of recreational activities such as fishing and walking, since TVA (Title Search, 1996) owned the shoreline property. The shoreline is the area of exposed sediments that result from raising and lowering the water level in the river. TVA obtained the shoreline property as flood easements before Watts Bar Reservoir was impounded (Title Search, 1996). The land area flooded consisted of 16,600 acres of bottomland and 14,200 acres of hill land (TVA, 1938). Since TVA owned the land and a large portion of bottomland had been lost; farming was probably not conducted adjacent to the river.

Fishing along the south bank of the Clinch River was popular for children, but adolescents and adults were needed to work on the farm the majority of the time (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996; Prichard, personal communication with C. Lewis, 1997). The tenants and their children also fished along the shoreline. Swimming was another popular recreational activity for the children, but this was primarily done in the creeks emptying into the Clinch River and not in the river itself (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). Fishing and swimming at Jones Island did not appear to be common practices, since the island was accessible only by boat or ferry, and children were the primary participants in fishing and swimming (Huber, personal communication with C. DaMassa, 1996; Waller, personal communication with C. Lewis, 1996). The ferry that went from the south bank of the river to Jones Island was often sunk during storms (Huber, personal communication with C. DaMassa, 1996).

Although children were occasionally found swimming in the Jones Island area, the river was not the primary body of water in which swimming occurred, and this pathway was not identified as significant in the screening analysis; therefore, the swimming pathway was not considered in the risk assessment. However, exposure to shoreline sediments was a possibility on the south side of the Clinch River in the Jones Island area due to tenants fishing and children playing near the water=s edge. Since there is evidence of exposure to shoreline sediments, this pathway is included in the risk assessment for the years following 1963 (see Section 7.2.1 for details).

7.2.2.3 External exposure from dredged sediment

External exposure from dredged sediment appears to be limited to those workers who disposed of the sediment. TVA purchased Jones Island from the last individual owner in January 1963. The dredging did not begin until October 1963; therefore, the possibility of private individuals being exposed to dredged spoils was greatly reduced, even though the last owner of the island was permitted to use the island until December 1963. Grubb Island was also used for the disposal of dredged material. However, this island was not used agriculturally before or after dredging. This pathway will not be considered further in the

exposure assessment because the likelihood of individuals other than workers being exposed to dredged spoils is low.

7.2.2.4 Ingestion of Meat and Milk

The only cattle raised on Jones Island were beef cattle (Huber, personal communication with C. DaMassa, 1996; Waller, personal communication with C. Lewis, 1996). The cattle grazed on pastures located on the island and drank from the river. The cattle also appear to have been fed grain (corn) and hay that were grown on the island (Huber, personal communication with C. DaMassa, 1996; Waller, personal communication with C. Lewis, 1996). No other animals were raised on the island (Waller, personal communication with C. Lewis, 1996).

The silage crops that were grown on the island were not irrigated with river water; rain was the only means of watering these crops (Waller, personal communication with C. Lewis, 1996). The land on the island was very fertile (sandy loam) and did not require additional fertilizer or topsoil (Huber, personal communication with C. DaMassa, 1996; Frankenburg, personal communication with W. Reed, 1996). Therefore, the only route of exposure resulted from the beef cattle drinking river water.

Hay was also grown along the south side of the river and was used as cattle feed. The land used for producing hay was not irrigated with river water (Waller, personal communication with C. Lewis, 1996). In addition, the soil used for crops was not enriched with river bottom sediments, including dredged spoils (Waller, Wade- personal communications with C. Lewis, 1996).

The beef cattle raised on the south side of the Clinch River also grazed on land near the river. Since several of the farms in this area bordered the river, the cattle had access to the river as a source of drinking water because fences were not erected to keep the cattle from the river, and no other boundaries existed between the river and private farmland (Waller, personal communication with C. Lewis, 1996). In addition, beef cattle require minimal upkeep and generally graze unattended. Even though river water was available to the beef cattle, these animals also had access to creeks, ponds, and springs (Waller, personal communication with C. Lewis, 1996). Dairy cattle, pigs, and chickens were also raised on the south side of the Clinch. However, these animals were usually maintained in fenced pastures and pens close to the family dwellings for ease in feeding, milking, and gathering eggs. These animals were fed grain and hay that were grown on the farm (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). Since the homes were not built on the river banks, these animals obtained water primarily from springs, creeks, and ponds located near the barn or from the family=s well (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996).

In some instances, both dairy and beef cattle were provided water from the family=s well or cistern (Wade, personal communication with C. Lewis, 1996). Even though the likelihood of dairy cattle obtaining river access is low, the drinking water exposure pathway will be considered in the risk assessment. The pathway is considered because these cattle were capable of escaping the fenced pastures to obtain water from the river. The drinking water pathway for beef cattle must be included due to the high probability that these

cattle used the river as a source of drinking water. This exposure pathway will be considered for the years following 1963 when Melton Hill Dam changed the water mixing patterns of the Clinch River (see Section 7.2.1 for details).

7.2.2.5 Ingestion of Produce Contaminated via Irrigation

The crops grown on Jones Island were used exclusively by the animals (Huber, personal communication with C. DaMassa, 1996; Waller, personal communication with C. Lewis, 1996). The silage crops that were grown on the island were not irrigated with river water; rain was the only means of watering these crops (Waller, personal communication with C. Lewis, 1996).

Homes in the Jones Island area were not built on the river banks due to early flooding (prior to Norris Dam construction) and because TVA owned all the shoreline on the south side of the river (USGS, 1941a; 1953a). In addition, several of the land grant farms did not have river frontage (Courthouse Retrieval Systems, Inc., 1996). The farms in this area were used primarily for raising beef cattle and growing silage crops (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996). Since the farmers in this area were cattlemen, the homes were not located on the river, and the crops were used only for silage, irrigation was not necessary along this stretch of the river. Therefore, the ingestion of produce contaminated via irrigation is not a relevant exposure pathway for radioactive substances released to the Clinch River.

7.2.2.6 Ingestion of Produce Contaminated via Dredged Sediment

There is no indication that produce for human consumption was grown on Jones Island at any time. There has also been no indication that vegetable gardens were placed adjacent to the river. This may be attributable to TVA=s ownership of the entire length of the shoreline and to the fact that no homes were built on the river. Vegetable gardens were generally planted close to the house for ease in obtaining the crop. In addition, farmers did not dredge the river bottom to enrich their soil in this area of the river (Waller, personal communication with C. Lewis, 1996). Therefore, the possibility of ingestion of produce grown on contaminated soil is small. Because no evidence has been obtained to support the use of dredged spoils for soil enrichment in this area, the ingestion of produce contaminated via dredged sediment cannot be considered a relevant exposure pathway for individuals inhabiting the Jones Island area of the Clinch River.

7.3 Grassy Creek Area - CRM 17.0 to CRM 14.0

The mouth of Grassy Creek is located on the north bank of the river at CRM 14.5, but the Grassy Creek area is considered to include the land adjacent to the banks of the Clinch River from CRM 17.0 to CRM 14.0 (Figure 7.1). The land areas in the vicinity of this creek have similar topographic features. The land is steeper on the south bank of the Clinch River in this area than in the Jones Island area, and fewer structures appear to have been built in this area (USGS, 1941b; 1953b).

7.3.1 Description of Land Use in the Grassy Creek Area

Grassy Creek is located on the north side of the Clinch River upstream from Bear Creek Road. This area is TVA land and is bordered on the northern side by U.S. Department of Energy (USDOE) property (Brown, 1996). Since this area is surrounded by government-owned property, access to this area is limited to government personnel. Therefore, the only way for the public to gain access to this area is by boat. The land lying along the south side of the Clinch River in this area was used for farming prior to 1949 (Brown, 1996). According to census information, there were 1,806 farms averaging 86.2 acres each in Roane County in 1954; by 1959 this number had shrunk to 992 farms averaging 96.0 acres each (TVA, 1963). However, TVA owned all the shoreline along this side of the river, so river accessibility may have been limited.

7.3.2 Potential Exposure Pathways in the Grassy Creek Area

The potential exposure pathways considered for this reach of the Clinch River include fish ingestion, external exposure from shoreline sediment, ingestion of meat and milk from cows, ingestion of produce contaminated via irrigation, and ingestion of produce contaminated via dredged sediment.

7.3.2.1 Fish Ingestion

TVA owned the north side of the Clinch River in the Grassy Creek area. This property was the proposed site for the Clinch River Breeder Reactor and was adjacent to DOE property, so access to this area was limited to government personnel. The shoreline was also owned by TVA, so fishing from the banks was prohibited.

Recreational fishing by boat could have occurred in this area. The shoreline on both sides of the river was owned by TVA, and there were very few structures in the area between CRM 17.0 and CRM 14.0 (USGS, 1941b; 1953b). The majority of the structures built in this reach of the river were located on the south side of the Clinch River in Bear Creek Valley, which lies between Pine and Chestnut Ridges, and Poplar Springs Valley, which lies between Chestnut and Dug Ridges (USGS, 1953b). Recreational fishing could have occurred in this area by boat or from the shore. Fishing in this area was done primarily by tenants using trotlines and Atangle lines@ (Prichard, personal communication with C. Lewis, 1997). These tenants used the fish as a source of protein for their families (Prichard, personal communication with C. Lewis, 1997). Land-owning farmers in this area had little time for recreational activities due to the time demands required for farming (Prichard, personal communication with C. Lewis, 1997). Since fishing occurred in this area, residents could have been exposed to contaminants released from White Oak Creek.

7.3.2.2 External Exposure from Shoreline Sediment

The greatest amount of external exposure from shoreline sediments occurs when the water level is low. Since access to the north side of the river in the Grassy Creek area was limited to government personnel and those traveling by boat, the probability of private individuals walking or swimming in this area is low.

The south side of the Clinch River has areas with very steep topography between CRM 17.0 and CRM 14.0, the areas adjacent to the Clinch River on Dug and Chestnut Ridges (USGS, 1941b; 1953b). The steep topography combined with the limited number of dwellings in this area indicates that the probability of recreational activities, such as shoreline walking or swimming, along this stretch of the river is low. However, since tenants living on farms on the south side of the river conducted fishing, exposure from shoreline sediments will be considered further for this area of the Clinch River.

7.3.2.3 Ingestion of Meat and Milk

The beef cattle raised on the south side of the Clinch River grazed on land near the river, but information obtained from the Roane County Courthouse records (Brown, 1996) and two sets of USGS maps (USGS, 1941b; 1953b), indicate that the number of farming tracts (plots of land designated as agricultural or as a farm on land title documents--deed, contract for sale or purchase of land, etc.) adjacent to the Clinch River and the number of dwellings in this area was small, possibly due to the topography. Even though few farms in this area had access to the river, the likelihood of the cattle using the river as a source of drinking water is high for this limited number of farms because no fences were erected between TVA flood easement property and private land. Hay was grown along this area of the river and was used as cattle feed. Dairy cattle were also raised in this section of Roane County. The dairy farms in this area averaged approximately 60 head of cattle, primarily Jerseys (Prichard, personal communication with C. Lewis, 1997). The cattle were kept near the barn for milking purposes; therefore, their access to the river would have been limited, but not completely restricted. Drinking water for these animals was a nearby stream, creek, pond, or spring, but if pasture was accessible along the river, river water was consumed. This exposure pathway will receive further consideration because dairy farms had river access in this area (Prichard, personal communication with C. Lewis, 1997). The drinking water pathway for beef cattle must also be included due to the high probability that these cattle also had river access.

7.3.2.4 Ingestion of Produce Contaminated via Irrigation

Farming was not conducted on the northern banks of the Clinch River because the property was government-owned (USDOE and TVA; USGS, 1941b; 1953b). The shoreline on the southern bank was also government owned (TVA; USGS, 1941b; 1953a). The land in this area was also steeper than the gently sloping lands found near Jones Island (USGS, 1941a; 1941b; 1953a; 1953b).

However, farming activities occurred on the land adjacent to the TVA property on the southern banks of the Clinch River (Prichard, personal communication with C. Lewis, 1997). Vegetable gardens were raised near the families= homes, which were located primarily in Bear Creek Valley and Poplar Springs Valley (Prichard, personal communication with C. Lewis, 1997). Both of these valleys have limited riverfront property (USGS, 1941b; 1953b). Because the land topography was steep and the amount of river frontage available to the farming families was limited, irrigation was not practiced in this area (Prichard, personal communication with C. Lewis, 1997). Since no evidence has been obtained to indicate the use of irrigation of produce in this area, the ingestion of produce contaminated via irrigation is not a relevant pathway for further consideration in this analysis of potential human exposure from past releases of radioactive materials into the Clinch River.

7.3.2.5 Ingestion of Produce Contaminated via Dredged Sediment

Farming activities occurred in this reach of the river, but vegetables were not grown adjacent to the river (Prichard, personal communication with C. Lewis, 1997). This may be attributable to TVA=s ownership of the entire shoreline, to the steep topography, and to the fact that no homes were built on the river. Vegetable gardens are generally planted close to the house for ease in obtaining the crop, but the number of houses appears to have been limited in this area (USGS, 1941b; 1953b). The majority of the structures were built on the south side of the Clinch River in Bear Creek Valley and Poplar Springs Valley, both of which have limited riverfront property (USGS, 1941b; 1953b). In addition, farmers did not dredge the river bottom to enrich the soil (Waller, personal communication with C. Lewis, 1996; Prichard, personal communication with C. Lewis, 1997). Therefore, the likelihood of ingestion of produce grown on contaminated soil in the Grassy Creek area is small. Because no evidence has been obtained to indicate the enrichment of soils with dredged spoils in this area, the ingestion of produce contaminated via dredged sediment is not considered relevant for further analysis.

7.4 K-25 Area - CRM 14.0 to CRM 5.0

The K-25 water intake, the location of primary interest in this segment of the river, is located on the north bank of the Clinch River at CRM 14.4, but the K-25 area includes the land adjacent to the banks of the Clinch River from CRM 14.0 to CRM 5.0 (Figure 7.1). The land areas further down river from the water intake have similar topographic features. The land is steeper on the banks of the Clinch River in this area, especially between CRM 8.0 and CRM 5.0, and few structures appear to have been built on either side of the river in this area (USGS, 1941b; 1953b). The majority of structures built on the north side of the Clinch River were located in Sugar Grove Valley, which is separated from the Clinch River by Black Oak Ridge (USGS, 1953b). On the south side of the river, the structures were concentrated around the communities of Union and Lawnville, both of which are located at the base of Pine Ridge and Lawnville Road, which connects these two communities (USGS, 1953b; 1968).

7.4.1 Description of the Land Use in the K-25 Area

Construction of the K-25 site, formerly known as the Oak Ridge Gaseous Diffusion Plant (ORGDP), was initiated in 1943 (ORNL, 1995). The mission of the facility was the enrichment of uranium via the gaseous diffusion process, which involved the movement of gaseous uranium hexafluoride across porous barriers (ORNL, 1995). The gaseous diffusion process at K-25 was operated until 1985, when the facility was placed on Aready standby,@ and was halted entirely in 1987 (ORNL, 1995).

7.4.2 Potential Exposure Pathways in the K-25 Area

The potential exposure pathways considered for this reach of the Clinch River include drinking water, fish ingestion, external exposure from shoreline sediment, ingestion of meat and milk from cows, ingestion of produce contaminated via irrigation, and ingestion of produce contaminated via dredged sediment.

7.4.2.1 Drinking Water

The water intake for the ORGDP or K-25 site is located at CRM 14.4. A filtration station is also located in this area, along with a water storage tank (USGS, 1953b). These three facilities together provide potable water for the industrial plant (Morton, 1963). The filters originally used in the filter station (Building K-1515) were sand, but these were changed in the 1970s to a combination of anthracite coal, sand, and granite (Bowman, personal communication with C. Lewis, 1997). The water storage tank is not located on the river, but is used to store the water for 2 1/2 to 3 days prior to use at the K-25 site (Bowman, personal communication with C. Lewis, 1997). K-25 is not the only location that uses this water source. Bear Creek Industrial Park also utilizes the K-25 water intake as a source of potable water (Bowman, personal communication with C. Lewis, 1997). This complex was built in the 1980s and has one company located within its boundaries, Scientific Ecology Group (SEG; Bowman, personal communication with C. Lewis, 1997). Building 9714, located 9 miles west of the Y-12 facility, also uses the K-25 water intake as a source of potable water (Bowman, personal communication with C. Lewis, 1997). Building 9714 serves as the garage for the Transportation and Safeguards Division (TSD; Joyce, personal communication with C. Lewis, 1998). Even though no information has been found to indicate that the K-25 water supply system was used by individuals living outside the Oak Ridge Reservation boundaries and the water was filtered prior to consumption, the water was consumed by individuals working at or visiting the K-25 site, Building 9714, or the Bear Creek Industrial Park; therefore, the drinking water pathway is relevant for further consideration in dose and risk estimates.

7.4.2.2 Fish Ingestion

The K-25 facility is located on the north side of the Clinch River north of the Oak Ridge Turnpike (Highway 58). Since this area is government-owned property, access to this area is limited to government personnel. Therefore, the only way for the public to gain access to this area is by boat. Fishing in this area is most likely a recreational practice as opposed to fishing for the purpose of providing the family with a major dietary source of protein.

The shoreline on the south bank of the river was owned by TVA, and the adjacent property was owned by U.S. DOE between CRM 14.0 and CRM 10.0 (Quitclaim Deed, 1987). In addition, there were very few structures in this area between CRM 14.0 and CRM 5.0 (see Section 7.4 for details; USGS, 1941b; 1953b). Therefore, fishing from the banks may not have been extensive in this area. Recreational fishing could have occurred in this area by boat or from the shore, but use of trotlines in this area has not been confirmed. Since fishing could have resulted in the exposure of residents to releases of radionuclides from White Oak Creek, the fish ingestion pathway will be included as a relevant exposure pathway for this section of the Clinch River.

7.4.2.3 External Exposure from Shoreline Sediment

The greatest amount of external exposure to sediment occurs when the water level is low. Since access to the north and the south banks of the Clinch River was limited to government personnel and between CRM 14.0 and CRM 10.0 to those traveling by boat, the probability of private individuals walking or swimming in this area is decreased substantially (intruders excluded). Between CRM 10.0 and CRM 5.0, the land topography is steep (USGS, 1941b; 1953b). In addition, very few structures were built outside the Sugar Grove Valley on the north side of the river or away from Lawnville Road on the south side of the Clinch River (USGS, 1941b; 1953b; Prichard, personal communication with C. Lewis, 1997). Since the topography is steep, the number of dwellings was limited, and access to the river was limited to a few trails and light duty roads between CRM 10.0 and CRM 5.0 (USGS, 1941b; 1953b), recreational activities such as shoreline walking and swimming along this section of the river were limited. Because this section of shoreline was government-owned, the topography was steep, and very few people lived in this area, the external exposure to shoreline sediments will not be considered further in the determination of exposure.

7.4.2.4 Ingestion of Meat and Milk

The beef cattle raised along the Clinch River between CRM 10.0 and CRM 5.0 grazed along the hillsides near the river (Prichard, personal communication with C. Lewis, 1997). The likelihood that these grazing beef cattle used the Clinch River as a source of drinking water in this area is high because no fences were erected to keep animals away from the river. Cattle grazing on the western tip of Black Oak Ridge, which is located on the north side of the river, may also have had access to the Emory River (USGS, 1941b; 1953b; 1968). Dairy cattle raised in this area were kept close to the barn or family dwelling for milking purposes; therefore, their access to the river was limited. Drinking water for the dairy cattle was a nearby stream, pond, or spring. Even though the dairy cattle were maintained close to the family dwelling and their likelihood of obtaining river water is limited, this exposure pathway will receive further consideration because these animals were not always contained in a barn or pasture. The drinking water pathway for beef cattle must be considered due to the high probability that these cattle had river access (Prichard, personal communication with C. Lewis, 1997).

7.4.2.5 Ingestion of Produce Contaminated via Irrigation

Farming activities occurred in this reach of the river, but crops were not grown adjacent to the river due to the topography and the location of the dwellings, which were concentrated in Sugar Grove Valley and along Lawnville Road (USGS, 1941b; 1953b; Prichard, personal communication with C. Lewis, 1997). Vegetable gardens were generally planted near the house for ease in obtaining the crop (Prichard, personal communication with C. Lewis, 1997). Irrigation was not used due to the location of the dwellings with respect to the river and the topography of the area (Prichard, personal communication with C. Lewis, 1997). Since farmers in this area did not irrigate, the ingestion of produce contaminated via irrigation is not included as a relevant exposure pathway for this stretch of the river.

7.4.2.6 Ingestion of Produce Contaminated via Dredged Sediment

Farms between CRM 14.0 and CRM 10.0 had no shoreline access because this property was owned by the Atomic Energy Agency (Quitclaim Deed, 1987). Shoreline property between CRM 10.0 and CRM 5.0 was difficult to access due to the limited number of roads to the river and the topography in this area. Because the area was difficult to access and heavy equipment was required to move large amounts of earth to the acreage designated for gardens, farmers did not dredge the river bottom to enrich their soil (Zirkle, personal communication with C. Lewis, 1996). Since garden plots were not enriched with river bottom soil, ingestion of produce contaminated via dredged sediment is not considered a relevant exposure pathway for this area.

7.5 Kingston Steam Plant Area - CRM 5.0 to CRM 2.0

The Kingston Steam Plant is located on the north bank of the river at CRM 2.6, but the Kingston Steam Plant area is considered to include the land adjacent to the banks of the Clinch River from CRM 5.0 to CRM 2.0 (Figure 7.1). The land areas bordering the river in the vicinity of this facility are easily accessible to the public and serve as popular recreational areas.

7.5.1 Description of Land Use in the Kingston Steam Plant Area

The area located on the peninsula formed by the Clinch and Emory Rivers (see Figure 7.1) was the site chosen by the TVA for construction of the Kingston Steam Plant (TVA, 1965). Originally four sites in Tennessee--Kingston, Louisville, Concord, and Lenoir City--were considered (TVA, 1965). The steam plant, located on Watts Bar Lake approximately two miles north of Kingston, Tennessee, was built between April 1951, when the original unit was installed, and December 1955, when the final unit went into operation (TVA, 1965). The installation covers approximately 800 acres and was constructed to provide electricity for Oak Ridge during production of Aatomic defense material@ (TVA, 1965). Until 1963, the Kingston Steam Plant was the world=s largest steam plant, with a generation capacity of 1,600,000 kW of electricity (TVA, 1965). Currently, the plant supplies electricity to homes and industries in East Tennessee (TVA, 1965).

7.5.2 Potential Exposure Pathways in the Kingston Steam Plant Area

The potential exposure pathways considered for this reach of the Clinch River include drinking water, fish ingestion, external exposure from shoreline sediment, ingestion of meat and milk from cows, ingestion of produce contaminated via irrigation, and ingestion of produce contaminated via dredged sediment.

7.5.2.1 Drinking Water

The Kingston municipal water supply was used by residents living within the city limits, but wells were used primarily by those individuals living outside the city (Zirkle, personal communication with C. Lewis, 1996; Mickey, personal communication with C. Lewis, 1996; Adkins, personal communication with C. Lewis, 1997). The residents living near the steam plant obtained water from cisterns and wells (Adkins, personal communication with C. Lewis, 1997). The Kingston Steam Plant obtained water supplies from the Clinch River at approximately CRM 3.0 for sanitary and industrial purposes, in addition to potable water (Morton, 1963).

The water treatment facility at the Kingston Steam Plant was built to supply the entire steam plant with domestic water and to soften the water for the boiler (TVA, 1965). The primary treatment of water was prechlorination (controls algae and disinfects water), chemical dosage (addition of aluminum sulfate, hydrated lime, and activated carbon), flocculation (mixing to settle out the particles), sedimentation (occurs when water passes through wooden baffles), and filtration (consists of filter media and porous plates) (TVA, 1965). Once the water goes through these initial phases, domestic water and boiler feedwater are separated. The domestic water is chlorinated, stabilized, and moved to storage tanks located in the powerhouse (TVA, 1965). Once the water reaches the storage tanks, it is ready for domestic use throughout the facility.

In December 1989, the Kingston Steam Plant began purchasing potable water from Midtown Utilities, which obtains its water from Rockwood Utilities (Sexton, personal communication with C. Lewis, 1997). The water supply for Rockwood Utilities is obtained from Rockwood Creek, a tributary to the Tennessee River (Ingram, personal communication with C. Lewis, 1997). Since Clinch River water was used as a source of drinking water for humans working at the steam plant, the drinking water pathway is considered for this location.

7.5.2.2 Fish Ingestion

Fishing along the Clinch River arm of the Watts Bar Reservoir has been a popular recreational activity, as well as a source of income for commercial fishermen (Stokes, personal communication with C. Lewis, 1996). However, the number of commercial fishermen is very small (approximately seven in five counties; Ebert, 1996). The area of the Clinch River near the Kingston Steam Plant is a popular location for winter recreational fishing because the warmer water attracts striped bass and catfish (Napier, personal communication with C. DaMassa, 1995). Fishing occurred in this area on the banks and from boats (Adkins, personal communication with C. Lewis, 1997). This section of the Clinch River (backwaters of

Watts Bar Reservoir) was accessible to the public (Ebert, 1996), so angler activity in this area is expected to be larger than further up the Clinch River (e.g., Jones Island). Since the Kingston Steam Plant was a popular location and easily accessible by land and by boat, the fishing pathway is included in the assessment of exposure, dose, and health risk for this area.

7.5.2.3 External Exposure from Shoreline Sediment

Fishing appears to be the most popular recreational activity in the vicinity of the steam plant (Adkins, personal communication with C. Lewis, 1997). However, walking and swimming could have also occurred in this location. Since the Kingston Steam Plant area was a popular location and easily accessible by land, the potential exposure from shoreline sediments is included as a relevant exposure pathway for further consideration in the dose reconstruction and assessment of risk.

7.5.2.4 Ingestion of Meat and Milk

Houses were located along both sides of the banks of the Clinch River in this area (USGS, 1941b; 1941c; 1953b). Prior to the impoundment of Watts Bar Reservoir, homes were built below the 741-foot elevation to access the river bottomland, which was the most fertile soil available (Zirkle, personal communication with C. Lewis, 1997). Once the reservoir was impounded, the land located below the 741 foot elevation level was flooded (USGS, 1941b; 1941c). The families in the affected area were forced to relocate upland, where farming activities continued. Dairy and beef cattle were raised in this area (Stokes, personal communication with C. Lewis, 1996; Adkins, personal communication with C. Lewis, 1997; Prichard, personal communication with C. Lewis, 1997). Since several homes remained near the river, even after the impoundment of Watts Bar Reservoir, accessibility of the river for both types of cattle is high. However, the cattle on the northern bank of the river had access not only to the Clinch River, but also to the Emory River and to Swan Pond, which is located on the western bank of the Emory River (USGS, 1941b). Since accessibility to the river was high for both types of cattle, the ingestion of beef and milk from cows is considered in the health risk assessment.

7.5.2.5 Ingestion of Produce Contaminated Via Dredged Sediment

Prior to the filling of Watts Bar Reservoir, irrigation was not necessary because the soil was very fertile bottomland (Zirkle, personal communication with C. Lewis, 1996). When bottomland was still available, approximately 1915-1940, corn was the primary crop grown in this area (Zirkle, personal communication with C. Lewis, 1996). However, when the reservoir flooded the bottomland, farmers residing below the 741-foot elevation level were relocated or forced to move upland to less fertile soil. However, farmers continued to raise vegetable gardens and to grow forage crops for their animals near the Kingston Steam Plant (Sparks, personal communication with C. Lewis, 1997; Prichard, personal communication with C. Lewis, 1997). The crops grown in this area included lettuce, green beans, cucumbers, tomatoes, and onions (Lowe, personal communication with C. Lewis, 1997). Fruits and berries were also grown in this area (Lowe, personal communication with C. Lewis, 1997). Crops that became the focus of farmers in this area were peaches, alfalfa hay, and soybean hay, because these crops could prosper on nutrient-deficient

soil (Zirkle, personal communication with C. Lewis, 1996). Despite the land=s infertility, no soil enrichment in garden plots was done since this would have required heavy equipment to move the earth from the river banks to the higher elevations (Zirkle, personal communication with C. Lewis, 1996). Since soil enrichment with dredged sediment was not necessary for producing crops in this area and since there is no evidence of the use of dredged sediment in agricultural or garden plots, this pathway is not considered further.

7.6 City of Kingston Area - CRM 2.0 to CRM 0.0

The City of Kingston is located between CRM 1.0 and CRM 0.0, but the City of Kingston area is considered to include the land adjacent to the banks of the Clinch River from CRM 2.0 to CRM 0.0 (Figure 7.1). Sections of the city are also adjacent to the Tennessee River, where water recreation is also popular.

7.6.1 Description of Land Use in the City of Kingston Area

The City of Kingston was established in 1799 as a result of a growing population around Fort Southwest Point (Carow, 1996). The fort served as an Army garrison between 1792 and 1807 and was surrounded by several trading posts, which helped to increase the population of Kingston (Carow, 1996). Kingston is located in Roane County, between Interstate 40 and Watts Bar Lake. The city served as the Tennessee state capital for one day on September 21, 1807, when the Tennessee House of Representatives convened (Carow, 1996).

7.6.2 Potential Exposure Pathways for the City of Kingston Area

The potential exposure pathways considered for this reach of the Clinch River include drinking water, fish ingestion, external exposure from shoreline sediment, and ingestion of meat and milk from cows.

7.6.2.1 Drinking Water

Prior to the impounding of Watts Bar Reservoir, the Kingston city water supply came from a spring located approximately 250 feet from the left (south) bank of the Clinch River (TVA, 1938). Since the spring would be flooded when the reservoir was filled, TVA proposed a series of wells as a new water supply. The well water would be pumped to an existing reservoir where the water would be chlorinated prior to use by the community (TVA, 1938). However, TVA did not drill the proposed wells. Instead, the water was, and currently is, obtained from a spring located near Midtown, Tennessee (Davis, personal communication with C. Lewis, 1997). This spring is gravity-fed and supplies approximately 150,000-180,000 gallons of water per day to the water treatment plant (Davis, personal communication with C. Lewis, 1997). In addition, in 1955, two water intakes were placed on the Tennessee River, just upstream from the confluence with the Clinch River at Tennessee River Mile (TRM) 568.4 (Davis, personal communication with C. Lewis, 1997). These intakes supply an additional 500,000-900,000 gallons of raw water per day to the water treatment plant in Kingston (Davis, personal communication with C. Lewis, 1997). Once the water (from the intakes and the spring) enters the treatment plant, the water passes through three filters (sand, charcoal, and one for

sediment removal) prior to distribution to the public (Davis, personal communication with C. Lewis, 1997). The plant provides approximately 580,000-700,000 gallons per day of treated water to the Kingston municipal water supply, while the remaining treated water is stored in tanks for later use (Davis, personal communication with C. Lewis, 1997).

If the Clinch River=s flow is greater than the flow of the Tennessee River, the water from the Clinch River travels up the Tennessee River (USDOE, 1996). The backflow results in Clinch River water entering the Kingston water intakes, and later reaching residents who obtain water from the Kingston municipal water supply system (Stokes, personal communication with C. Lewis, 1996; USDOE, 1996). The residents living within the city limits use the municipal water supply, but those living outside the city (Zirkle, personal communication with C. Lewis, 1996; Mickey, personal communication with C. Lewis, 1996; Adkins, personal communication with C. Lewis, 1997) use wells. Since evidence of the use of Clinch River water as a source of drinking water for people in the Kingston area has been obtained, this pathway is included as a relevant pathway in the exposure assessment.

7.6.2.2 Fish Ingestion

Fishing along the Clinch River has been a popular recreational activity, as well as a source of income for commercial fishermen in the Kingston area (Stokes, personal communication with C. Lewis, 1996). Since Kingston is very close to the Tennessee River, fishermen could use both rivers for recreational (Zirkle, personal communication with C. Lewis, 1996; Mickey, personal communication with C. Lewis, 1996; Adkins, personal communication with C. Lewis, 1997) and commercial fishing (Hargis, 1968). It is unlikely that the Clinch River was commercially fished to a large extent due to the limited access for larger boats and the proximity of the Watts Bar commercial fishery (Ebert, 1996). The number of full-time commercial fishermen in Watts Bar Reservoir is also very small (Ebert, 1996). Hargis (1968) indicated that in 1967, a total of seven full-time commercial fishermen operated in Rhea, Meigs, Roane, Anderson, and Loudon counties combined.

Recreational fishing has been conducted on the Clinch River and in Watts Bar Reservoir since it was impounded (Ebert, 1996). A practice known as Afish fries,@ communities cooking large quantities of fish, was very popular in this area (Evans, personal communication with C. DaMassa, 1995; Stokes, personal communication with C. Lewis, 1996). Trotlines were also used in this area (Zirkle, personal communication with C. Lewis, 1996; Adkins, personal communication with C. Lewis, 1997). Because fishing is such an integral part of life in Kingston (Stokes, personal communication with C. Lewis, 1996; Zirkle, personal communication with C. Lewis, 1996; Mickey, personal communication with C. Lewis, 1996; Adkins, personal communication with C. Lewis, 1997), the fish ingestion pathway is considered further in the evaluation of this location.

7.6.2.3 External Exposure from Shoreline Sediment

After World War II, residential subdivisions were developed on the outskirts of Kingston, as well as on the northwest shore of Watts Bar Reservoir (TVA, 1987). The increased building in this area would have increased the amount of recreational activity that occurred on the river during this time period. Swimming, walking, fishing, wading for mussels, and collecting of freshwater pearls occurred along the banks of the Clinch River in the Kingston area (Stokes, personal communication with C. Lewis, 1996; Zirkle, personal communication with C. Lewis, 1996; Mickey, personal communication with C. Lewis, 1996). Since most residents participated in recreational activities that occurred on the banks of the Clinch River, the shoreline exposure pathway is considered relevant in the dose reconstruction at this location.

7.6.2.4 Ingestion of Meat and Milk

Since several of the farms in this area bordered the river, the possibility of both beef and dairy cattle having access to the river as a source of drinking water is high. Hay was grown along this area of the river and was used as cattle feed for both beef and dairy cattle. Even though milk was easily obtainable from White Stores and other smaller grocers (Adkins, personal communication with C. Lewis, 1997) and from the local dairies--Brogdan=s Dairy, John Bacon Dairy, Suddath Dairy (Mickey, personal communication with C. Lewis, 1996; Lowe, personal communication with C. Lewis, 1997), and Norris Creamery (Stokes, personal communication with C. Lewis, 1996), this exposure pathway receives further consideration, due to the high probability that both dairy and beef cattle had river access.

7.7 Summary of Exposure Pathways

Overall, the pathways considered along the Clinch River from Jones Island to Kingston included drinking water, fish ingestion, external exposure from shoreline sediment, exposure from dredged sediment, ingestion of meat and milk from cows, and ingestion of produce contaminated via dredged sediment. For each reach of the river, the pathways to be considered were identified by a conservative screening analysis (see Section 3). Demographic information was then used to determine if the individual pathways actually existed. Table 7.1 lists all pathways considered and those that will be included in the risk assessment, along with the rationale behind each decision. The potential exposure pathways to be included in the risk assessment for the Jones Island area include fish ingestion, external exposure from shoreline sediment, and ingestion of meat and milk from cows. The Grassy Creek and the K-25 areas were considered to be one area for the fish, meat, and milk ingestion pathways. However, external exposure from shoreline sediment was considered only for the Grassy Creek area, due to the occurrence of tenant fishing along the Clinch River in this area. The drinking water exposure pathway was included only for the K-25 area, since the water intake supplies water to K-25, one outlying building within the Y-12 complex, and a private business. Pathways of concern for the Kingston Steam Plant area include drinking water, fish ingestion, external exposure from shoreline sediment, and ingestion of meat and milk from cows. Pathways that are considered in the risk assessment for the City of Kingston include drinking water, fish ingestion, external exposure from shoreline sediment, and ingestion of meat and milk from cows that consumed river water.

7.8 Target Individuals and Exposure Parameters

Target individuals for the Task 4 assessment are persons exposed or potentially exposed to radionuclides released from the Oak Ridge National Laboratory (ORNL) via White Oak Creek. The target individual may be exposed to all routes (pathways) of exposure (i.e., drinking water, fish ingestion, etc.) or to a combination of various exposure routes or pathways. The populations of interest or target individuals representative of these populations must be identified prior to the determination of all exposure parameters.

7.8.1 Water Ingestion

The water ingestion pathway has two target individuals: adults and children. For the two water intakes serving industrial facilities (K-25 and the Kingston Steam Plant), children were not considered because they were not generally taken to the K-25 facility, the TSD garage at the Y-12 facility, Scientific Ecology Group (SEG), or the Kingston Steam Plant, all of which are serviced by water drawn from the Clinch River (Bowman, personal communication with C. Lewis, 1997). For the Task 4 analysis, visitors at these facilities were the target adults addressed because government employees (DOE) were not specified as endpoints in this assessment.

The third water intake of interest, the Kingston Water Treatment Facility, has multiple target individuals, including adults (resident or transient) and children. Resident adults lived and worked within the city limits of Kingston. Most, if not all, of their water was obtained from the Kingston Municipal Water Supply system. Transient individuals were adults who lived in the City of Kingston, but who worked in another city or outside the city limits or vice versa–they did not spend all their time in Kingston and obtained only a fraction of their water from the Kingston municipal water supply. The children considered are residents of Kingston or attended schools within the city limits. The water ingestion pathway exposure parameters are provided in Table 7.2.

Table 7.1 Summary of potential exposure pathways considered for the Clinch River, based on land use information and a screening analysis.

Location: Jones Island Area - CRM 21.0-17.0

Pathways Considered	Pathways To Be Included	Rationale
Drinking water	No	River water was not used as a source of drinking water. Potable water was obtained from wells.
Fish ingestion	Yes	Tenants and children fished from riverbanks and used trotlines.
External exposure from shoreline sediment	Yes	Fishing occurred and children played along the riverbanks in this area. Pathway is considered for the years following 1963 due to the complete mixing of the Clinch River with releases from White Oak Creek after the installation of Melton Hill Dam.
External exposure from dredged sediment	No	Dredging occurred after TVA purchased the island.
Swimming	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of meat and milk	Yes	Beef cattle grazed on land adjacent to the river and drank river water, if accessible. This pathway is considered after 1963 due to complete mixing of releases with water in the Clinch River in the Jones Island area.
Ingestion of produce contaminated via irrigation	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of produce contaminated via dredged sediment	No	Soil enrichment with dredged spoils did not occur in this area due to the location of the dwellings away from the river and the difficulty associated with transporting dredged sediment to garden plots.

Table 7.1 (Continued)

Location: Grassy Creek Area - CRM 17.0-14.0

Pathways Considered	Pathways To Be Included	Rationale
Drinking water	No	River water was not used as a source of drinking water. Potable water was obtained from cisterns and wells.
Fish ingestion	Yes	Tenants fished on the banks and used trotlines.
External exposure from shoreline sediment	Yes	Tenant families fished from the riverbanks.
External exposure from dredged sediment	No	Dredging was not conducted in this area.
Swimming	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of meat and milk	Yes	Beef and dairy cattle were raised on farms in this area and had access to the river.
Ingestion of produce contaminated via irrigation	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of produce contaminated via dredged sediment	No	Soil enrichment with dredged spoils was not practiced due to the topography and the family dwellings being located away from the river.

Table 7.1 (Continued)

Location: K-25 Area - CRM 14.0-5.0

Pathways Considered	Pathways To Be Included	Rationale
Drinking water	Yes	Potable water for K-25, a private company, and one building within the Y-12 complex was obtained from the river, but filtered prior to use.
Fish ingestion	Yes	Recreational fishing by boat may have been conducted; the results will be identical to those for the Grassy Creek area.
External exposure from shoreline sediment	No	Government-owned land limited access to shoreline and riverbanks. Steep topography, limited access roads, and minimal number of dwellings limited resident shoreline use.
External exposure from dredged sediment	No	No dredging occurred in this reach of the river.
Swimming	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of meat and milk	Yes	Beef cattle grazed the hillsides adjacent to the river. Dairy cattle were kept in pastures close to the barn for milking purposes; therefore, less access to the river.
Ingestion of produce contaminated via irrigation	No	Pathway was eliminated in the screening analysis (see Section 3.0).

Table 7.1 (Continued)

Location: Kingston Steam Plant Area - CRM 5.0-2.0

Pathways Considered	Pathways To Be Included	Rationale
Drinking water	Yes	Potable drinking water for the steam plant is obtained from the Clinch River, but filtered prior to use.
Fish ingestion	Yes	Popular fishing location along the Clinch River.
External exposure from shoreline sediment	Yes	Land in this area provides easy access for recreational activities.
External exposure from dredged sediment	No	No dredging occurred in this area.
Swimming	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of meat and milk	Yes	Cattle farming occurred along the banks of the Clinch River in this area.
Ingestion of produce contaminated via irrigation	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of produce contaminated via dredged sediment	No	Pathway was eliminated in the screening analysis (see Section 3.0).

Table 7.1 (Continued)

Location: Kingston Area - CRM 2.0-0.0

Pathways Considered	Pathways To Be Included	Rationale
Drinking water	Yes	Potable water supply was obtained from an underground spring and from the Tennessee River, which could receive Clinch River water under certain flow conditions.
Fish ingestion	Yes	Fishing was a popular recreational activity in the City of Kingston.
External exposure from shoreline sediment	Yes	Recreational water activities were conducted frequently in this area.
External exposure from dredged sediment	No	Dredging did not occur in this area.
Swimming	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of beef and milk	Yes	Beef and dairy cattle grazed on land near the river.
Ingestion of produce contaminated via irrigation	No	Pathway was eliminated in the screening analysis (see Section 3.0).
Ingestion of produce contaminated via dredged sediment	No	Soil enrichment with dredged spoils was not practiced due to the effort involved in moving large amounts of earth to designated garden plots.

Table 7.2 Values for Parameters Used in the Water Ingestion Pathway.

Parameter	Symbol	Range	Mean	Distribution	Rationale
Average daily consumption of drinking water by adults (L d^{-1})	$U_{water, adult}$	0.8-2.4	1.6	Uniform	This range was used for residents of Kingston (within the city limits) and for visitors at K-25, at the Kingston Steam Plant, and in the City of Kingston. The total fluid intake of an adult under normal conditions ranges between 1.0 and 2.4 L d^{-1} (ICRP, 1975). The consumption of tap water and water-based drinks that include tea, coffee, soft drinks, and alcoholic beverages ranges between 0.4 and 2.2 L d^{-1} (ICRP, 1975). Tea, lemonade, and coffee are popular beverages in the Clinch River area (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996; Adkins, personal communication with C. Lewis, 1997). A uniform distribution was chosen to account for persons who drink primarily carbonated beverages or milk and persons who consume primarily water and water-based beverages.
Fraction of contaminated water consumed by adults at the industrial facilities (unitless)	$F_{cwk, adult}$	0.2-0.6	0.4	Uniform	This range was used for K-25, TSD garage, SEG, and Kingston Steam Plant visitors. The range considers the consumption of other fluids that were not obtained from a contaminated source, such as carbonated drinks. The range also considers that individuals at these sites were also obtaining water from other locations that may not have been contaminated (e.g., bottled water). The mean (0.4) was selected to allow for the majority of fluids to be consumed at home with meals. A uniform distribution was chosen because individuals at these locations had several beverage options.

Table 7.2 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Fraction of contaminated water consumed by adults (transient individuals) (unitless)	$F_{cw,\ transient}$	0.2-0.6	0.4	Uniform	This range was used for the individuals who consumed water provided by the Kingston municipal water supply system, but were not full time residents of Kingston. These individuals could be people who worked in Kingston, but resided outside the city limits or in another town or vice versa. A uniform distribution was chosen to account for the uncertainty in the fraction of contaminated water consumed by mobile receptors.
Fraction of contaminated water consumed by adults (resident individuals) (unitless)	$F_{cw,\ resident}$	0.2-0.8	0.5	Uniform	The range was used for the adults who were full time residents of Kingston (individuals who worked and resided within the city limits). These receptors could conceivably obtain 100% of their total fluid intake from the Kingston City Municipal Water Supply. However, with the availability of other beverages (soft drinks, milk, beer, etc.), the range accounts for consumption of other types of fluids. A uniform distribution was chosen because various combinations of nonwater-based beverages are available to adults.

Table 7.2 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Average daily consumption of drinking water by infants (L d^{-1})	$U_{water, infant}$	0.12-0.95	0.54	Uniform	The range was applied to children consuming water from the municipal water supply system in the City of Kingston, as children were not located at K-25 or the Kingston Steam Plant. The upper bound is considered to be representative of a child that was formula-fed (dry powder mixed with water). The lower bound is representative of a child who consumes small amounts of water or water-diluted juice within the first year of life. The lower bound indicates that a child receiving diluted juice would not consume less than 4-oz. per day (averaged over a one-year period). The upper bound indicates that a child consuming water-based formula would consume no more than 32 oz. per day (eight 4-oz. servings; Eisenberg et al., 1994; Mount, personal communication with C. Lewis, 1997). The range provided for infants is consistent with values provided in Rupp (1980) and NCRP (1984). A uniform distribution was selected to represent the range within which all values for the true but unknown intakes of drinking water (averaged over the first year of life) have equal probability.

Table 7.2 (continued)

Parameter	Symbol	Range	[Median]	Distribution	Rationale
Average daily consumption of drinking water by children ages 1-4 years old (L d^{-1})	$U_{water,\ 1\text{-}4\ yr.\ old.}$	0.24-0.95	[0.50]	Derived	The range applied to children (males and females) using the Kingston municipal water supply system between the ages of 1 and 4 years was derived from age-dependent milk consumption rates (see Table 7.4) and the assumption that milk constitutes 50-60% of a 1-4 year old child's total fluid intake. The derived distribution is a combination of a uniform distribution for the percentage of the total fluid that is milk (a) and the distribution for the milk consumption rate ($U_{milk,\ 1\text{-}4\ yrs.\ old}$). The equation used to derive the water ingestion distribution is $U_{water,\ 1\text{-}4\ yr.\ old} = U_{milk,\ 1\text{-}4\ yr.\ old} \times (1\text{-}a)/a$. The lower bound is equivalent to approximately one 8 oz. glass, while the upper bound is equivalent to approximately four 8-oz. glasses. The recommended fluid intake (all fluids, including those received from foods) for children ages 1-3 is between 1.4 and 1.7 L d^{-1} (based on an average weight of 13 kg; Mount, personal communication with C. Lewis, 1997). The range provided is consistent with the recommended total fluid intake rate when the combined total of tap water and water-based drinks (tea, Kool-Aid, lemonade, etc.) provided here and the milk consumption rates for this age group (see Table 7.4) are summed (excluding water received from carbonated beverages and food). For the case when the child is consuming both contaminated water and contaminated milk, the ingestion rates for each fluid are determined independently. The derived distribution (subjective probability distribution for the total fluid intake minus the subjective probability distribution for the amount of fluid that is milk) was selected to describe the uncertainty associated with the consumption rate of drinking water by children between the ages of 1and 4.

Table 7.2 (continued)

Parameter	Symbol	Range	Median] or Mean	Distribution	Rationale
Average daily consumption of drinking water by children ages 5-14 years old (L d^{-1})	$U_{water,\ 5\text{-}14\ yr.\ old.}$	0.24-1.2	[0.77]	Derived	The age-dependent ingestion rate of drinking water by children between the ages of 5 and 14 was derived as the difference between the total fluid intake of milk, tap water, and water-based beverages (ICRP, 1975) and age-dependent milk consumption rates (see Table 7.4). The upper bound is equivalent to approximately five 8-oz. glasses, the lower bound is equivalent to one 8-oz. glass, and the median is equivalent to three 8-oz. glasses of water or water-based beverages per day. The range provided is consistent with the total fluid intake for children between the ages of 5 and 14 to remain hydrated (Campbell, personal communication with C. Lewis, 1997, with body weights provided in Burmaster and Crouch, 1997).
Fraction of contaminated water consumed by children (unitless)	$F_{cw,\ child}$	0.2-0.8	0.5	Uniform	The range applies to children living in the City of Kingston, as drinking water was not obtained for children from the Clinch River upstream from Kingston. The range indicates that at least 20% of the drinking water was obtained from the municipal water supply system, which is potentially contaminated by the Clinch River. A uniform distribution was chosen because children living outside the city limits, but attending city schools, would not obtain all their water from the municipal water supply, as did children residing and attending schools in the City of Kingston.

Table 7.2 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Fraction of radionuclides attached to the sediment remaining after water treatment at the K-25 facility, the Kingston Steam Plant, and the Kingston Water Treatment facility (unitless)	F_{PRR}	0.05-0.15	0.1	Uniform	The radionuclides that readily attach to the sediment include the isotopes of cesium and cobalt. These can be removed from the water prior to filtering in the settling basin or during the filtering process. The radionuclides can be removed in these two processes because the raw water is filtered and gently stirred to force the sediment out of its dissolved or suspended state. Studies conducted with similar treatment processes reported removal of between 84 and 99% of ^{137}Cs and ^{60}Co (Lindsten et al., 1965). The amount of radioactive material and the amount of sediment in the water determine the effectiveness of the removal of radionuclides attached to sediment. A uniform distribution was used to describe the true but unknown fraction of radionuclides attached to sediment removed by water treatment processes.
Fraction of radioactive cations remaining in water after treatment at the K-25 facility, Kingston Steam Plant, and the Kingston Water Treatment facility (unitless)	F_C	0.72-0.92	0.82	Uniform	Ruthenium and strontium are radioactive cations, do not readily react with sediment, and are also beta emitters. These cations do not bind readily to particles unless an anionic polymer is added to the water. A study was conducted at the ORGDP water treatment facility to determine the amount of beta radioactivity removed from the raw water. From 1955 to 1964, the amount of beta radioactivity was reduced 8-28% at the 95% confidence limit (Schultz, 1966). Studies conducted with similar treatment processes reported removal of ^{90}Sr up to 15% (Lindsten et al., 1965). The water treatment processes are similar at all three locations of interest on the lower Clinch River. A uniform distribution was selected to describe the unknown removal effectiveness, which depends on the amount of radioactive material in the water and on the amount of polymer added.

Table 7.2 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Fraction of radioactive anions remaining in water after treatment at the K-25 facility, the Kingston Steam Plant, and the Kingston Water Treatment Facility (unitless)	F_A	0.22-0.75	0.49	Uniform	Iodine is a radioactive anion that does not readily bind to particles unless a cationic polymer is added to the water. At the K-25 water treatment plant, aluminum sulfate (no form specified) is added to the raw water; liquid aluminum sulfate is added at the Kingston Steam Plant; and Ultrion-8156 is added at the Kingston Water Treatment facility. Aluminum sulfate and Ultrion-8156 are cationic coagulants and cause the anions present in the water to bind to the metal cations present in the coagulant. Once bound, the anions will settle out in the floc. Studies conducted with similar treatment processes reported removal of ^{131}I between 25 and 78% (Straub et al., 1951). Since the removal depends on the amount of radionuclides in the water and the amount of coagulant added, a uniform distribution was chosen to account for the uncertainty in this parameter.
Fraction of Clinch River water backflowing up the Tennessee River to the Kingston water intakes (unitless)	B_{RK}	0.02-0.17	0.08	Uniform	Modeling efforts by TVA indicate that backflow occurs approximately 2 hrs d^{-1} (0.08), which was considered to be the mean. This range indicates that backflow up to the Kingston water intakes occurs at least 0.5 hr d^{-1}, but no more than 4 hrs d^{-1}. Since backflow is a result of power generation, a uniform distribution was chosen to describe the true but unknown value of the fraction of time that Clinch River water backflows up the Tennessee River to the Kingston water intakes.

7.8.2 Fish Ingestion

For the fish ingestion pathway, three target individuals were identified for Task 4: Category 1, 2, and 3 fish consumers. Category 1 fish consumers eat fish frequently (one to two and one half meals per week). Examples of these individuals include tenant farmers and commercial fishermen. Although commercial fishermen were considered to be Category 1 fish consumers, commercial fishing on the Clinch River was not likely due to the limited access for larger boats and the proximity of the Watts Bar Reservoir commercial fishery (Ebert, 1996). Therefore, the number of commercial fishermen on this body of water was small. However, tenants lived on farms located adjacent to the Clinch River and fished frequently with trotlines (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996; Prichard, personal communication with C. Lewis, 1997). Category 1 fish consumers could be either male or female, with consumption rates dependent on body weight (see Table 7.3 for specific consumption rates) and the number of meals consumed.

Categories 2 and 3 were used to describe the variety of patterns exhibited by fish consumers who do not use fish as a major source of protein, including recreational anglers. With the accessibility of multiple bodies of water in the area (Watts Bar Reservoir, the Emory River, and the Tennessee River), recreational anglers could have used a combination of locations to obtain fish.

Category 2 fish consumers eat fish less frequently than a Category 1 consumer, but more often than a Category 3 consumer. Examples of a Category 2 consumer include recreational anglers who consume a portion of all fish caught, but do not fish solely in the Clinch River, neighbors of recreational anglers who share in the anglers catch, and wealthy land owners who used other sources of protein (e.g., Jones Island area land owners). A Category 2 fish consumer can be either male or female, with consumption rates dependent on body weight and the number of meals consumed. Table 7.3 provides the exposure parameters for the fish ingestion pathway.

Category 3 fish consumers are individuals who fish for sport or consume a limited amount of fish. These individuals are similar to the Category 2 consumer, with the exception that they do not consume fish as often. Category 3 fish consumers also visit a variety of fishing locations: Watts Bar, the Emory River, the Tennessee River, and lakes and streams in the area. This type of fish consumer can also be male or female, with consumption being dependent on body weight and the number of meals consumed.

7.8.3 Milk and Meat Ingestion

The milk and meat ingestion pathways also have multiple target individuals. For milk ingestion, the target individuals are two groups of children (less than 6 years of age or 6 years of age and older) and adults. The fraction of contaminated milk consumed is based on the child's age and the situations encountered at that age. Children less than 6 years of age may have obtained all their milk from a backyard cow. Children in this age group are more likely to remain at home the majority of the time

Table 7.3 Values for Parameters Used in the Fish Ingestion Pathway.

Parameter	Symbol	Range	Mean	Distribution	Rationale
Number of fish meals per week for a Category 1 fish consumer (meals $week^{-1}$)	$N_{fish,\ Category\ 1}$	1.0-2.5	1.8	Uniform	The range is provided for a Category 1 fish consumer, who is described as an individual (male or female) who eats fish frequently (commercial fishermen or tenant farmers). Category 1 fish consumers eat between 1 and 2.5 fish meals per week. Tenant farmers along the Clinch River were known to have consumed fish on a regular basis, but also used pork, chicken, and beef to feed their family. The upper bound of the range is consistent for farmers who depended on fish as a major source of protein. A subsistence fisherman consumes a maximum of 180 g d^{-1} (EPA, 1989; McCormack and Cleverly, 1990, as cited in Ebert, 1996) or 1260 g wk^{-1}. Using an average portion size of 200 g $meal^{-1}$, the maximum number of fish meals per week is approximately 6, which is greater then the maximum for tenant farmers. Rupp et al. (1980) indicated that a maximum fish consumer in the East South Central region of the United States (including Tennessee) consumes approximately 2.4 meals per week (using a 200 g per meal portion size). The lower bound is consistent with the 99th percentile fish consumer for the East South Central region of the US (0.96 meals wk^{-1} using 200 g $meal^{-1}$; Rupp et al. 1980) and with the consumption of the most popular fish species found in Watts Bar Reservoir (29 g d^{-1} or 203 g wk^{-1}; Ebert, 1996). A uniform distribution was selected to encompass those tenant farmers whose lifestyle and meat availability fluctuated.

Table 7.3 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Number of fish meals per week for a Category 2 fish consumer (meals $week^{-1}$)	$N_{fish, Category\ 2}$	0.25-1.3	0.78	Uniform	The range is provided for a Category 2 fish consumers (male or female), who is described as an individual who consumes fish, but does not consume only the fish caught in the Clinch River (recreational anglers). Category 2 consumers eat between 1 fish meal per month and a little more than 1 fish meal per week. The Category 2 fish consumer could be a recreational or sport fisherman, who was often found in the Kingston area and at the Kingston Steam Plant. The upper bound represents individuals who consumed any available fish species found in Watts Bar Reservoir (Ebert, 1996) and the 99th percentile fish consumer located in the East South Central region of the US (Rupp et al., 1980). The lower bound represents those who consumed only the most popularly harvested species (largemouth bass, channel catfish, white crappie, and white bass) and those who consumed only one species (catfish, white crappie, or white bass; Ebert, 1996). The lower bound is also representative of the 90th percentile fish consumer found in the East South Central region of the United States (Rupp et al., 1980). A uniform distribution was selected to account for the variation in fish consumption patterns exhibited by various recreational anglers and Category 2 fish consumers.

Table 7.3 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Number of fish meals per week for a Category 3 fish consumer (meals $week^{-1}$)	$N_{fish, Category\ 3}$	0.04-0.33	0.19	Uniform	The range is provided for a Category 3 fish consumer (male or female), who is described as an individual who consumes fish infrequently and does not consume fish solely from the Clinch River. Category 3 consumers consume between 1 fish meal every 6 months and 1 fish meal every 3 weeks. The Category 3 consumer can be compared to a purely sport fisherman, who could be found in the Kingston area and at the Kingston Steam Plant. The upper bound represents those individuals who fish when time permits or occasionally consume fish. The lower bound represents the individual who fishes primarily for sport and gives the fish to others or does not keep the fish for food. The mean value is indicative of the national average (Ebert, 1996). The lower bound is consistent with the average fish consumer found in the East South Central region of the US (Rupp et al., 1980). The uncertainty within this group was described with a uniform distribution to encompass the characteristics of sport fishermen and Category 3 fish consumers.
Portion size of a fish meal for males (kg $meal^{-1}$)	$P_{fish, male}$	0.10-0.30	0.20	Uniform	The range is provided for a male fisherman who consumed between 100 and 300 grams of fish per meal. The mean value (200 g) is the average portion size determined by Rupp et al. (1980). The upper bound is equivalent to 11 ounces of fish per meal and the lower bound is equivalent to 4 ounces. A uniform distribution was chosen to account for the uncertainty associated with fish portion size among the male population.

Table 7.3 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Portion size of a fish meal for females (kg meal^{-1})	$P_{fish, female}$	0.08-0.25	0.17	Uniform	This range is for a female fisherman. These values are based on the premise that consumption rates are generally proportional to body weight (female body weight 58 kg; male body weight 70 kg; Ebert, 1997). The lower bound represents 3 ounces of fish and the upper bound 9 ounces per meal. A uniform distribution was chosen to encompass finicky consumers as well as hearty consumers.
Fraction of fish consumed that is contaminated (unitless)	F_{cf}	0.2-1.0	0.6	Uniform	The range provided assumes that the fish consumed by individuals was obtained from the contaminated body of water as well as from noncontaminated sources. Tenant farmers used trotlines solely in the Clinch River, near Jones Island. Since the tenant farmer is a target individual, the maximum was determined to be 1.0. The mean value is consistent with the ratio for the number of river fishing trips to lake fishing trips for the average Tennessee angler (0.6; Ebert, 1996). The lower bound is representative of individuals in the Kingston area who utilize the fishing on Watts Bar Reservoir, the Tennessee River, the Emory River, the Clinch River, and the numerous lakes in the area to obtain fish. A uniform distribution was selected to describe the uncertainty associated with the true, but unknown fraction of contaminated fish consumed.
Fraction of radionuclide remaining in fish after food processing (unitless)	$F_{r, fish}$	0.8-0.9	0.85	Uniform	The range provided assumes that 80-90% of the ^{90}Sr and ^{137}Cs remains in the fish after frying (IAEA, 1992), which was considered the typical method of preparing fish. Since the fraction of radionuclide remaining in the fish is dependent on the radionuclide of interest, the true but unknown value was represented by a uniform distribution.

and to get a significant part of their nutrition from milk. Children 6 years of age and older may have obtained a percentage of their milk from a noncontaminated source since they were likely to have obtained milk from school during nine months of the year (Dreicer et al., 1990; Downen, 1955; 1956). Adults were considered to have been able to obtain milk solely from a backyard cow or from a noncontaminated source.

The meat ingestion pathway has two primary target individuals (adults and children), but the fraction of contaminated meat consumed varies with location along the Clinch River. If an individual was living or obtaining meat from the area between Jones Island and K-25, the likelihood exists that the individual was obtaining 100% of his or her meat from animals that drank Clinch River water. The area between Jones Island and K-25 was heavily farmed (beef cattle), and residents in this area consumed farm-raised meat primarily, due to the scarcity of commercially processed meat (Waller, personal communication with C. Lewis, 1996; Wade, personal communication with C. Lewis, 1996; Prichard, personal communication with C. Lewis, 1997). Individuals residing in Kingston and surrounding areas had access to commercially processed meat, in addition to locally raised meat (Adkins, personal communication with C. Lewis, 1996; Stokes, personal communication with C. Lewis, 1996). The exposure parameters used in the milk and meat ingestion pathways are provided in Table 7.4.

7.8.4 External Exposure

The target individuals for the external exposure pathway are adults or children who utilized the shoreline when the water level was low and the sediments were exposed. The amount of sediment exposed depends on the location of interest. The widths of the exposed shoreline sediments near the Kingston Steam Plant and the City of Kingston are different than the width of the shoreline near Jones Island and K-25 (see Section 10). Individuals who utilized the riverbanks for fishing or walking were also exposed to sediments when the water level was up. The exposure to shoreline sediments decreases when the water level is up because the water acts as a shielding material. The exposure parameters for the external pathway are provided in Table 7.5.

Table 7.4 Values for Parameters Used in the Milk/Meat Ingestion Pathway.

Parameter	Symbol	Range	Mean or (Mode)	Distribution	Rationale
Average daily consumption of milk by adults (L d^{-1})	$U_{milk, adult}$	0.24-0.46	0.35	Uniform	This range is consistent with values reported by the USDA (1980), Dreicer et al. (1990), Cole and Ronning (1974), and Rupp (1980). The values are approximately equivalent to one-half (4 **2** oz.) and one glass (8 **2** oz.) of milk per day, respectively. The Clinch River area was an area in which milk could be obtained from a home-raised animal or from a local dairy; therefore, milk was readily available to most individuals. This range also allows for adults having access to other fluids than milk (e.g., water, coffee, soft drinks, and tea). A uniform distribution was selected to account for the uncertainty associated with beverage preference and annualized consumption rate of milk.
Fraction of milk consumed by adults that is contaminated (unitless)	$F_{cm, adult}$	0.63-1.0	(1.0)	Triangular	This range is representative of milk production in the area of Anderson, Loudon, Roane, Blount, and Knox counties, which was not adequate too completely satisfy the needs of these communities. According to NCI (1997) these areas were milk deficient. Locally produced and potentially contaminated milk constituted approximately 63% of the total needs (Hoffman et al., 1996). The lower bound is representative of the fact that the area was milk-deficient and that commercial milk brought in from other areas constituted 37% of the total amount of noncontaminated milk for this area. The upper bound is indicative of an adult who consumed milk from a backyard cow. A triangular distribution was chosen because the mode (1.0) was the value most likely to represent the actual milk consumed by an adult owning a backyard cow.

Table 7.4 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Average daily consumption of milk by infants (L d^{-1})	$U_{milk, infant}$	0.12-0.98	0.54	Uniform	This range represents the milk consumption rates of a newborn to a twelve-month-old. During the first few months of life, babies are fed every 3 to 4 hours, with each serving consisting of 4-6 oz. of milk (Eisenberg et al., 1994). As the baby develops, the number of feedings decreases, but the amount consumed increases (Eisenberg et al., 1994; Mount, personal communication with C. Lewis, 1997). An infant who is being breast fed, but is occasionally provided fresh cow's milk, will consume no less than 4 oz. of fresh cow's milk per day, which is the lower bound (Mount, personal communication with C. Lewis, 1997). "The number of infants consuming breast milk decreases as a function of age, while the number of infants consuming fluid cow's milk increases continuously" (NCI, 1997). An infant who is being fed fresh cow' s milk will consume no more than 32 oz. of cow's milk per day. The upper bound is based on eight 4-oz. feedings per day (feeding every 3 hours). The average value represents an infant who consumes approximately one-half breast milk and one-half fresh cow's milk (four and one-half 4-oz. servings as an infant or two 8-oz. servings as an older infant, who is usually given juice in addition to milk). The range is consistent with values reported by USDA (1965), Rupp (1980), Dreicer et al. (1990), Simon et al. (1990), and NCI (1997). A uniform distribution was chosen to represent the range within which all values for the true but unknown intake of fresh cow's milk (averaged over the first year of life) have equal probability.

Table 7.4 (continued)

Parameter	Symbol	Range	(Mode)	Distribution	Rationale
Average daily consumption of milk by children ages 1-4 years old (L d^{-1})	$U_{milk,\ 1\text{-}4\ yrs.}$	0.24-1.2	(0.53)	Triangular	This range is for milk drinkers and is based on information provided by the National Cancer Institute (NCI, 1997). This range includes females, who consume less fluid milk on average than males (Dreicer et al., 1990). According to the NCI study, 73% of the male and 71% of the female milk drinkers within the ages of 1-4 years fall within this range (NCI, 1997). The NCI (1997) study concluded that approximately 17% of children (both genders) between the ages of 1 and 4 did not consume milk. Pao and Burke (1975) determined that a maximum value of 1.33 L d^{-1} is exceeded by fewer than 2.5% of children aged 3-11 years (Rupp, 1980). The lower bound is indicative of approximately one 8-oz. glass of milk per day, with less than 6% of the population drinking less than one glass of milk (NCI, 1997). The upper bound indicates at most 5 glasses (8-oz. each) of milk are consumed per day. The mode is consistent with the per capita consumption rate for rural farm residents (no age specified) and the median per capita milk consumption rate for milk drinkers in the State of Tennessee in 1954 (NCI, 1997). The range is also consistent with values reported by USDA (1965), Rupp (1980), Dreicer et al. (1990), and Simon et al. (1990). A triangular distribution with a mode of 0.53 was chosen to represent the uncertainty associated with the daily consumption rate of milk (averaged over a four-year period) by children aged 1-4 years old.

Table 7.4 (continued)

Parameter Symbol	Symbol	Range	(Mode)	Distribution	Rationale
Average daily consumption of milk by children ages 5-9 years old (L d^{-1})	$U_{milk,\ 5\text{-}9\ yrs.}$	0.24-1.2	(0.76)	Triangular	This range encompasses 69% of males and 65% of females who consume milk (NCI, 1997). An average of 22% of this population group do not consume milk (NCI, 1997). The lower bound indicates children between the ages of 5 and 9 who drink approximately one 8-oz. glass of milk per day. The mode is the median per capita consumption rate for milk drinkers in the State of Tennessee in 1954 for this age group (NCI, 1997). The upper bound indicates at most 5 glasses (8-oz. each) of milk are consumed per day. The range is also consistent with values provided in the literature (USDA, 1965; Rupp, 1980; Dreicer et al., 1990; Simon et al., 1990). A triangular distribution was selected to represent the uncertainty associated with the true but unknown value of the average daily consumption rate for children between the ages of 5 and 9 years.
Average daily consumption of milk by children ages 10-14 years old (L d^{-1})	$U_{milk,\ 10\text{-}14\ yrs.}$	0.24-1.2	(0.8)	Triangular	This range includes 61% of males and 57% of females who consume milk (NCI, 1997). An average of 29% of this age group does not consume fresh fluid cow's milk (NCI, 1997). The range indicates children between the ages of 10 and 14 who consume approximately one 8-oz. glass of milk per day, but no more than five 8-oz. glasses of milk per day. The mode is the median consumption rate of milk drinkers in the State of Tennessee in 1954 for children aged 10-14 years (NCI, 1997). The range is consistent with values reported by USDA (1965), Rupp (1980), Dreicer et al. (1990), and Simon et al. (1990). A triangular distribution was chosen to describe the uncertainty associated with the average daily consumption rate for children aged 10-14 years who lived along the Clinch River.

Table 7.4 (continued)

Parameter	Symbol	Range	(Mode)	Distribution	Rationale
Fraction of milk consumed by children under 6 years old that is obtained from local sources (unitless)	$F_{cm, child < 6}$	0.63-1.0	(1.0)	Triangular	This range indicates that children under the age of 6 obtain most of their milk at home (e.g., a backyard cow or commercially produced). Since the commercially produced milk for the area constituted only 63% of the total required, 0.63 was chosen as the lower bound for this age group (Hoffman et al., 1996). The upper bound indicates the dominant source of milk that was obtained from a backyard cow. A triangular distribution was chosen because the mode (1.0) was the value most likely to represent the fraction of milk consumed by children under the age of 6 that was obtained from a backyard cow.
Fraction of milk consumed by children over 6 years old that is obtained from local sources (unitless)	$F_{cm, child > 6}$	0.5-1.0	(0.8)	Triangular	This range indicates that children over the age of 6 could receive half of their milk from noncontaminated sources. School-aged children received milk for lunch as part of the School Milk Program (Dreicer et al., 1990: Downen, 1955; 1956). The upper bound indicates the possibility that children could have obtained all their milk from a backyard cow. A triangular distribution was chosen because the mode (0.8) was the most likely value to represent the fraction of milk consumed by children older than 6 who obtained milk from a commercial source and a backyard cow.

Table 7.4 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Average daily consumption of meat by adults (kg d^{-1})	$U_{meat, adult}$	0.15-0.3	0.23	Uniform	This range is consistent with the values reported by ATSDR (1992), USDA (1944), and Rupp (1980). The upper bound is based on the average total intake of meat (0.258 kg d^{-1}) for adults (Rupp, 1980). This maximum value is indicative of an individual who raised most of the meat consumed (e.g., rural individuals living along the Clinch River). The lower bound is representative of individuals who obtained a mixture of farm-raised and store-bought meat (e.g., residents of the City of Kingston). A uniform distribution was used to describe the meat ingestion rate due to the uncertainty in the average annual consumption of meat by males and females.
Fraction of meat consumed by residents living between or near Jones Island and K-25 that is obtained from local sources (unitless)	$F_{cb, J-K}$	0.3-0.8	0.55	Uniform	This range is representative of adults who lived between Jones Island and K-25, who raised most of the meat consumed on their farms. Beef, pork, and chickens were raised for family consumption along this stretch of the Clinch River. Meat was not regularly purchased from the grocery store in this area. Since the cattle were the only animals using river water, the only meat contaminated was the beef. This range indicates that beef was consumed 30% of the time, but no more than 80%. Since a variety of meat was available to individuals in this area, a uniform distribution was selected to represent the true but unknown fraction of meat contaminated that was consumed.

Table 7.4 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Fraction of meat consumed by residents living in the City of Kingston that is obtained from local sources (unitless)	$F_{cb,\ Kingston}$	0.1-0.6	0.35	Uniform	This range is representative of adults who lived in the City of Kingston, who raised their own animals or obtained a portion of their meat from local farms. Beef, pork, and chickens were raised for family consumption along this stretch of the Clinch River; however, many grocery stores were convenient to this area, and meat was regularly purchased from the grocery store. In this area, the cattle were the only animals using river water; therefore, the only meat contaminated was beef. The range indicates that at least 10% of the beef consumed was contaminated, but no more than 60% due to the many places to buy and the various types available. A uniform distribution was chosen to represent the uncertainty in the fraction of contaminated meat consumed by various individuals who have different tastes.
Fraction of radionuclide remaining in meat after food preparation (unitless)	$F_{r,\ meat}$	0.2-0.9	0.55	Uniform	This range indicates that 20-90% of the radionuclide remains in the meat after cooking (IAEA, 1992). The range is consistent for various methods of preparing meat: baking, boiling, frying, roasting, grilling, or mincing for cow, pig, deer, bird, or rabbit. The fraction of radioactive material remaining in the food after preparation has been described by a uniform distribution to include all the methods used to cook meat. (See Appendix 7B.)

Table 7.4 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Delay time between milking and consumption of milk (d)	D_{milk}	0.33-1.5	0.92	Uniform	This range represents a delay time from 8 hours to 1 days. Limited refrigeration was available until the late 1950s, so milk was stored in springs and creeks until it was consumed. A uniform distribution was chosen to account for the individuals using milk obtained in the morning later the same day and those individuals storing the milk for later use.
Daily intake rate of water for dairy cattle ($L\ d^{-1}$)	$Q_{m,\ dairy}$	32-60	46	Uniform	This range represents the watering habits of backyard dairy cattle. Dairy cattle used for commercial milking are given foods that increase their need for water (Neel, personal communication with W. Reed, 1997). The commercial cows are also generally younger, which is associated with a greater water intake (Neel, personal communication with W. Reed, 1997). However, for backyard cows living along the Clinch River, hay and grass were the only foods available. Since these cows were kept and utilized until their death, their water intakes could range dramatically. Under these conditions, a dairy cow was expected to consume at least 32 $L\ d^{-1}$, but no more than 60 $L\ d^{-1}$ over its lifetime, with the average being around 46 $L\ d^{-1}$ (Neel, personal communication with W. Reed, 1997). This range is also consistent with the values reported by McKone (1988), Cole and Ronning (1974), and the NCRP (1984). A uniform distribution was selected to represent the uncertainty in the annual average daily water intake by dairy cows caused by their age and feeding regime.

Table 7.4 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Daily intake rate of water for beef cattle ($L\ d^{-1}$)	$Q_{f,\ beef}$	25-55	40	Uniform	This range represents the watering habits of beef cattle. Beef cattle used in commercial production are given foods that increase their weight, which also increases water intake. The beef cows living along the Clinch River were provided hay, some grain, and grass. No additional supplements were added. Under normal conditions, the beef cow was expected to consume at least 25 $L\ d^{-1}$, but no more than 55 $L\ d^{-1}$ over its lifetime, with the average being around 40 $L\ d^{-1}$. (McKone, 1988; Cole and Ronning, 1974; and NCRP, 1984). This range is also consistent with the fact that beef cattle require at least 10% less water than dairy cattle (Neel, personal communication with W. Reed, 1997). A uniform distribution was selected to represent the uncertainty in the average annual daily water intake by beef cows caused by their feeding regime.
Fraction of contaminated water consumed by dairy cows (unitless)	P_m	0.05-0.25	0.15	Uniform	This range indicates that dairy cattle did not have frequent access to river water. Dairy cattle are kept near the barn and in fenced pasture areas for ease in milking (Waller, personal communication with C. Lewis, 1996). These animals obtained the majority of their water from springs, streams, ponds, and creeks (Waller, personal communication with C. Lewis, 1996). Since these animals occasionally escaped from the fenced areas, a uniform distribution was selected to represent the uncertainty in the annual average value for this parameter.

Table 7.4 (continued)

Parameter	Symbol	Range	Mean	Distribution	Rationale
Fraction of contaminated water consumed by beef cattle (unitless)	P_f	0.25-1.0	0.63	Uniform	This range reflects the fact that beef cattle were allowed to graze freely. The river was the largest source of water for these animals, especially those on Jones Island who exclusively drank river water. The river was not fenced off; therefore, the beef cattle had relatively easy access. However, there were creeks, streams, ponds, and springs also available to these animals. Since the beef cattle had a variety of water sources, a uniform distribution was selected to represent the fraction of water consumed that was contaminated.

Table 7.5 Parameter Values in the External Exposure Pathway

Parameter	Symbol	Range	Mean	Distribution	Rationale
Fraction of days per year using the shoreline when the water level is low near the Kingston Steam Plant and Kingston (unitless)	$F_{t(down),\ K}$	0.07-0.1	0.085	Uniform	This range reflects the fact that the shoreline in this area contains contaminated sediments that are exposed when the water level is low. The exposed shoreline is shallow, with a broad apron at low water levels. The range is equivalent to an individual (possibly an angler) using the shoreline for 24-36 d yr^{-1} in the winter and spring. The lower bound assumes an individual is exposed 1 day each week (4 per month) during the winter months (3) and the spring months (3). The upper bound is equivalent to 1.5 days per week for 6 months of the year. A uniform distribution was selected to account for the uncertainty associated with shoreline use.
Fraction of days per year using the shoreline when the water level is low along the Clinch River (unitless)	$F_{t(down),\ CR}$	0.02-0.03	0.025	Uniform	This range reflects the fact that the shoreline in other areas along the Clinch was not as accessible, did not have as broad an apron, or was not used as frequently as the areas near the Kingston Steam Plant and the City of Kingston. The range is equivalent to an individual using the shoreline for 6-12 d yr^{-1} during the winter and spring. The lower bound represents exposure to the shoreline 1 day per month during the winter (3 months) and spring (3 months). The upper bound is 2 days per month during the winter and spring. Individual preferences determine the amount of time spent on the shoreline; therefore, a uniform distribution was chosen to represent the true but unknown fraction of time that an individual uses the shoreline when the water level is low along the Clinch River.

Table 7.5 (continued)

Parameter	Symbol	Range	Mean or (Mode)	Distribution	Rationale
Fraction of days per year using the shoreline when the water level is high along the entire length of the Clinch River (unitless)	$F_{f(up)}$	0.07-0.20	0.14	Uniform	This range indicates that a variety of recreational activities, such as walking and fishing, were conducted on the banks of the Clinch River, especially in the summer in the Kingston area. The maximum value represents 72 d yr^{-1}, which is equivalent to 3 days per week (12 days per month) during the spring and summer months (6 months). The lower bound is equivalent to 1 day per week for 6 months of the year. The mean value is 2 days per week for recreational activities along the shoreline when the water is up. A uniform distribution was chosen to represent uncertainty due to differences in shoreline use among various individuals.
Fraction of hours per day using the shoreline for recreational activities during the spring and summer (unitless)	ET_{ss}	0.02-0.33	(0.17)	Triangular	This range indicates that a variety of recreational activities were conducted on the banks of the Clinch River during the spring and summer months. The maximum value indicates that 8 hours per day could be spent fishing or participating in other recreational activities along the shoreline. The minimum indicates that at least one-half hour was spent along the shoreline during recreational activities. A triangular distribution with a mode of 4 hours was selected to represent the uncertainty associated with the number of hours per day spent along the shoreline.
Fraction of hours per day using the shoreline for recreational activities during the fall and winter (unitless)	ET_{fw}	0.02-0.21	(0.08)	Triangular	This range describes the amount of time spent per day along the shoreline in the fall and winter. The maximum was expected to be no greater than 5 hrs d^{-1}, while the minimum was expected to be no less than one-half hour per day. The mode (2 hrs. d^{-1}) of a triangular distribution was chosen to represent the average amount of time spent participating in recreational activities along the shoreline during the fall and winter.

7.9 Population Estimates for Potentially Exposed Individuals

The number of individuals exposed or potentially exposed is another important element of a risk assessment. Each exposure pathway should be considered with respect to the size and type of populations that may be affected by the pathway (e.g., farmers, tenants, fishermen, residents, nonresidents, and visitors). Exposed populations should be identified as accurately as possible. Estimates have been provided as a range of the number of distinct individuals potentially exposed over the time period of interest via the following pathways: drinking water, fish ingestion, milk ingestion, meat ingestion, and external exposure. The estimates cover the entire 50-year period, unless otherwise stated for each pathway.

7.9.1 K-25 Water Intake

The population exposed to the water ingestion pathway varies for each intake. The water intake at the K-25 facility serves the K-25 facility in addition to Building 9714 at the Y-12 facility and the Scientific Ecology Group (SEG) building in the Bear Creek Industrial Park (Bowman, personal communication with C. Lewis, 1997). The number of individuals potentially exposed via this intake is estimated to be no more than 1,000, since government personnel employed at K-25 and Y-12 are not considered. No children are included in this estimate because they were not expected to visit this location.

7.9.2 Kingston Steam Plant Water Intake

The water intake at the Kingston Steam Plant served approximately 400 employees in 1997 (Proctor, personal communication with C. Lewis, 1997). This intake was installed in 1954 and used until December 1989, at which time the drinking water source was changed to the Midtown Utility District (Sexton, personal communication with C. Lewis, 1997). After 1989, no further exposures via this pathway occurred at this location. The greatest number of employees at the Kingston Steam Plant was about 3,000 during the construction years (Proctor, personal communication with C. Lewis, 1997). The smallest number of individuals employed at this site occurred in 1997 (375 employees; Proctor, personal communication with C. Lewis, 1997). When the drinking water was first purchased from another supplier, approximately 900 employees were located at the Kingston Steam Plant (Proctor, personal communication with C. Lewis, 1997). Since drinking water was drawn from the Clinch River for 35 years and the number of employees was decreasing over those years, the estimated number of potentially exposed individuals is 300-3,000. No children were included in this estimate because they were not employed at the facility or expected to be visitors at this location.

7.9.3 Kingston Water Intake

The Kingston Municipal Water Supply serves 3,090 people as of 1997 (Ladd, personal communication with C. Lewis, 1997). The number of individuals served by the municipal water system has increased over the years, but has remained relatively constant during the past few years (Ladd, personal communication

with C. Lewis, 1997). Since the population served by the municipal water supply has increased since 1940, the estimated number of potentially exposed individuals is 800-4,000 persons. The lower population estimate is based on census data from 1940, which indicated a population of approximately 900 individuals living within the city limits of Kingston (Vickers and Cunningham, 1996). The upper population estimate is based on 1990 census data for the number of residents living within the city limits of Kingston (approximately 4,600), many of whom still use wells to obtain their drinking water (Vickers and Cunningham, 1996). Since the mean household size has varied between 2 and 4 since 1950 (Ebert, 1996), the estimated number of potentially exposed children is 200-2,000. Table 7.6 provides population data for the City of Kingston between 1940 and 1990.

Table 7.6 Population in the City of Kingston, Tennessee, from 1940 until 1990 (USDOC, 1963; Vickers and Cunningham, 1996).

Year	Population
1940	900
1950	1,600
1960	2,000
1970	4,100
1980	4,600
1990	4,600

7.9.4 Category 1 Fish Consumers

The population potentially exposed to the fish ingestion pathway consists of Category 1, 2, and 3 fish consumers. Category 1 consumers for the Clinch River were considered to be primarily tenant farmers and their families. Tenant farmers were assumed to consume fish as a necessity to provide the proper nutrition for themselves and their families. Other individuals may have consumed large quantities of fish out of preference for this type of food and not out of necessity. According to census data for 1940 and 1950, the number of tenants in Anderson, Loudon, and Roane counties decreased dramatically over this time period (USDOC, 1942; 1952). Table 7.7 provides the farm populations for these three counties, which were chosen due to their proximity to the Clinch River. Since the tenant farming population decreased during the ten-year period, the estimated number of potentially exposed individuals is fewer than 400 persons, which includes individuals who preferentially consumed fish. The population estimate is based primarily on census data from 1950, which indicated that a population of 100, 100, and 200 tenant farming families lived in Anderson, Loudon, and Roane counties, respectively (USDOC, 1942; 1952). Since the mean household size varied between 3 and 4 during the 1950s and 1960s (Ebert, 1996), the estimated number of potentially exposed children is less than 150.

Table 7.7 Farm Populations for Anderson, Loudon, and Roane Counties (USDOC, 1942).

County	Year	Owners	Tenants
Anderson	1940	1,400	400
	1950	800	100
Loudon	1940	1,000	300
	1950	600	100
Roane	1940	1,100	400
	1950	900	200

7.9.5 Category 2 and 3 Fish Consumers

Category 2 and 3 fish consumers for the Clinch River were considered to be primarily recreational anglers and their families. However, there are no data available to provide estimates for the number of individuals who used the Clinch River recreationally (Ebert, 1996). Table 7.8 gives the total estimated angler population for Anderson, Loudon, Meigs, Rhea, and Roane counties (counties chosen because of their proximity to the Clinch River) and the estimated total number of potentially exposed individuals based on family size (Ebert, 1996). Since the fishing population can fluctuate from year to year, the estimated number of potentially exposed individuals is 30,000-100,000 persons (Ebert, 1996). The lower population estimate is based on the total population from the five counties in 1950, the estimated number of anglers, and the mean household size for this area. The estimated number of anglers is derived from information concerning the number of individuals who fished lakes, rivers, or streams (Ebert, 1996). The upper population estimate is based on the total population for each of the five counties of interest (Ebert, 1996). The estimated number of potentially exposed children is 5,000-20,000.

Table 7.8 Estimated Number of Potentially Exposed Recreational Anglers and Their Families.

Year	Estimated Number of Anglers	Anglers and Their Families[a]
1950	8,200	32,800
1960	8,600	32,300
1970	8,800	28,200
1980	10,100	28,300
1990	10,700	27,800

[a] based on the mean household sizes of 4.0, 3.8, 3.2, 2.8 and 2.60 persons per family in 1950, 1960, 1970, 1980, and 1990, respectively.

7.9.6 Farms in the Jones Island Area

The milk and meat exposure pathways generally have the same exposure populations since most rural farms in the Anderson, Loudon, and Roane county area had both beef and dairy cattle (Waller, personal communication with C. Lewis, 1996; Stokes, personal communication with C. Lewis, 1996; Prichard, personal communication with C. Lewis, 1997). For the Jones Island reach of the Clinch River, beef cattle were the primary concern because dairy cattle had limited access to the river water (Waller, personal communication with C. Lewis, 1996). Approximately twenty parcels of land were located in the Jones Island area adjacent to the Clinch River in Loudon and Roane Counties (Courthouse Retrieval System, Inc., 1996). In 1950, the rural populations were approximately 14,500 and 18,200 in Loudon and Roane counties, respectively (USDOC, 1963), and the number of farms in these counties was approximately 1,500 and 1,600, respectively (USDOC, 1952). Using this information, the average number of persons per farm was approximately 10, including tenants. The closest urban area was Lenoir City, which had a population of approximately 5,200 in 1950 (USDOC, 1963). Local meat (beef) and milk were not sold commercially in this area due to the lack of refrigeration (Waller, personal communication with C. Lewis, 1996). The estimated number of exposed individuals for the Jones Island reach of the river is 50-500 individuals. Since the mean household size varied between 2 and 4 during the period of operation of the Oak Ridge Reservation (Ebert, 1996), the estimated number of potentially exposed children is 25-100.

7.9.7 Farms in the Grassy Creek Area

For the Grassy Creek/K-25 reach of the Clinch River, beef and dairy cattle were of concern because each farm had its own cattle to produce meat and milk for the residents (Prichard, personal communication with C. Lewis, 1997). Approximately thirty-five parcels of land designated for farming or other agricultural activities were located in the Grassy Creek/K-25 area adjacent to the Clinch River in Roane County (both sides of the river; Courthouse Retrieval System, Inc., 1996). In 1950, the rural population was approximately 18,200 in Roane county (USDOC, 1963). The number of farms in this county was approximately 1,600 (USDOC, 1952). Using this information, the average number of persons per farm was approximately 11, including tenants. The estimated number of exposed individuals for the Grassy Creek reach of the river is 50-600 individuals. Since the mean household size varied between 2 and 4 during the operation of the Oak Ridge Reservation (Ebert, 1996), the estimated number of potentially exposed children is 25-100.

7.9.8 Farms in the Kingston Steam Plant and Kingston Areas

For the Kingston Steam Plant area and the rural areas of Kingston, beef and dairy cattle were of concern because each farm had its own cattle to produce meat and milk for the residents (Stokes, personal communication with C. Lewis, 1996). Approximately eighteen parcels of agricultural land were located in the Kingston Steam Plant/Kingston area adjacent to the Clinch River in Roane County (both sides of the river; Courthouse Retrieval System, Inc., 1996). In 1950, the rural population was approximately 18,200 in Roane County (USDOC, 1963), and the number of farms was approximately 1,600 (USDOC, 1952). Using this information, the average number of persons per farm was approximately 11, including tenants.

The estimated number of exposed individuals in the rural setting for the Kingston Steam Plant - Kingston reaches of the river is 50-500 individuals. Since the mean household size varied between 2 and 4 during the operation of the Oak Ridge Reservation (Ebert, 1996), the estimated number of potentially exposed children in the rural setting is 25-100.

7.9.9 Urban Residents of Kingston

Since the City of Kingston was located on the Clinch River and individuals living in the city limits had access to farm-raised milk and meat, the urban setting was also considered. In 1950, the population within the city limits of Kingston was approximately 1,600 persons (USDOC, 1963), growing to approximately 4,600 by 1990 (Vickers and Cunningham, 1996). Since these individuals could have had access to locally raised milk and meat, an additional 200-2,000 individuals could have been exposed over fifty years of operations at the Oak Ridge Reservation. An additional 50-500 children could have been potentially exposed who lived in the City of Kingston (based on the average number of individuals per household, 2-4). Table 7.9 provides the populations of Anderson, Loudon, and Roane counties from 1940-1990.

7.9.10 Fall/Winter Recreation Participants

The external exposure pathway has two different scenarios: water is lowered in the fall and winter and the sediments are exposed, and the water level is up in the spring and summer and acts as a shielding material. The populations potentially exposed to the unshielded shoreline sediment consist of individuals fishing on the banks of the river and individuals using the shoreline for other activities, such as walking. The fishing population consists primarily of adults when the water level is low. Due to cooler temperatures, children do not normally fish in the winter months. The total number of adults who walk during low water levels is also reduced because of the cooler temperatures. The estimated number of potentially exposed adults when the water is low is 3,000-10,000. This range is based on 10% of the estimated number of recreational anglers using the Clinch River in the fall and winter (Ebert, 1996). The estimated number of potentially exposed children utilizing the shoreline of the river is 300-1,000.

7.9.11 Spring/Summer Recreation Participants

The populations potentially exposed to the unshielded shoreline sediment when the water is raised consist of individuals fishing on the banks of the river as well as individuals using the shoreline for other activities, such as walking. The fishing populations consist of both adults and children. Children normally fish more often in the spring and summer months. The number of individuals who walk also increases in the warmer temperatures and longer days. The estimated number of potentially exposed adults when the water is high is 15,000-50,000, a range based on 50% of the estimated number of recreational anglers using the Clinch River in the spring and summer (Ebert, 1996) and the population of Roane County (Vickers and Cunningham, 1996; USDOC, 1943). The estimated number of potentially exposed children utilizing the shoreline of the river is 1,500-5,000.

Table 7.9 Populations in Anderson, Loudon, and Roane counties (Vickers and Cunningham, 1996; USDOC, 1943).

Location	Year	Population
Anderson County	1940	26,500
	1950	59,400
	1960	60,000
	1970	60,300
	1980	67,300
	1990	68,300
Loudon County	1940	19,800
	1950	23,200
	1960	23,800
	1970	24,300
	1980	28,600
	1990	31,300
Roane County	1940	27,800
	1950	31,700
	1960	39,100
	1970	38,900
	1980	48,400
	1990	47,200

7.9.12 Summary of Potentially Exposed Populations

The potential exposure pathways for the Clinch River are drinking water, fish ingestion, milk and meat ingestion, and external exposure. Each of these pathways and the locations along the river has different potentially exposed populations. The estimated range for the number of distinct individuals potentially exposed for each pathway and location is provided for the time periods of concern. Table 7.10 shows the potential exposure pathways, locations, time period of interest, and the estimated number of individuals for each pathway and location.

Table 7.10 Potential exposure pathways, locations of interest, applicable time periods, and estimated exposure populations for the Clinch River.

Exposure Pathway	**Location**	**Applicable Time Period**	**Population**	
			Adult	Child
Drinking water	K-25	1944-1991	< 1,000	0
	Kingston Steam Plant	1954-1989	300-3,000	0
	Kingston	1955-1991	800-4,000	200-2,000
Fish ingestion (Category 1)	Clinch River	1944-1991	< 400	< 150
Fish ingestion (Category 2 and 3)	Clinch River	1944-1991	30,000-100,000	5,000-20,000
Milk and/or meat ingestion	Jones Island	1963-1991	50-500	25-100
	K-25/Grassy Creek	1944-1991	50-600	25-100
	Kingston Steam Plant/Kingston	1944-1991	50-500	25-100
External exposure (water lowered)	Clinch River	1944-1991	3,000-10,000	300-1,000
External exposure (water raised)	Clinch River	1944-1991	15,000-50,000	1,500-5,000

7.10 Demography Summary

The risk assessment conducted for Task 4 extensively utilized the demography of areas surrounding the Oak Ridge Reservation to determine the risk to individuals in these locations. Demographic information was used to identify the locations of interest, the exposure pathways, the target individuals, and the exposed populations. The five reaches of interest on the Clinch River were Jones Island (CRM 21.0–17.0), Grassy Creek (CRM 17.0-14.0), K-25 (CRM 14.0-5.0), Kingston Steam Plant (CRM 5.0-2.0), and the City of Kingston (CRM 2.0-0.0). The pathways of exposure considered in the risk assessment included drinking water, fish ingestion, external exposure from shoreline sediments, and ingestion of milk or meat from cattle. The target individuals for the risk assessment were adults and children who consumed water, milk, fish, or meat, or who used the shoreline of the river for recreational purposes. The estimated exposed populations consisted of individuals participating in the various activities that could have potentially resulted in an exposure to radionuclides released from Oak Ridge National Laboratory via White Oak Creek.

The fish ingestion pathway and the external exposure pathway were analyzed for the entire reach of the Clinch River (CRM 21.0 - CRM 0.0). Target individuals for the fish ingestion pathway were identified as Category 1 (e.g., tenant farmers), Category 2, and Category 3 (both recreational anglers). The exposed population due to fish ingestion was estimated for all five reaches of the Clinch River (combined). The estimate for the Category 1 fish consumers from CRM 21.0 to CRM 0.0 was <400 adults and <150 children. For the external exposure pathway, the population exposure estimates were also for all five reaches of the Clinch River (combined). The adult population was estimated to be between 3,000-10,000 when the water was low and between 15,000-50,000 when the water was raised. The child population estimates were 300-1,000 and 1,500-5,000, respectively.

For the Jones Island area, the pathways of interest were fish ingestion (previously discussed), external exposure from shoreline sediments (previously discussed), and ingestion of milk or meat. For the milk and meat ingestion pathways, the exposed population estimates in the Jones Island area were between 50-500 adults and between 25-100 children. The estimated exposed children were not divided into age groups due to limited availability of age-specific census information for Tennessee.

The Grassy Creek area and K-25 had slightly different pathways of interest: The Grassy Creek area exposure pathways included fish ingestion (previously discussed), external exposure to shoreline sediment (previously discussed), and ingestion of milk or meat; exposure pathways for K-25 were drinking water, fish ingestion (previously discussed), and ingestion of milk or meat. However, the estimates for the exposed populations in these two areas were combined due to their close proximity. The drinking water pathway for K-25 was estimated to be fewer than 1,000 individuals (adults only), since AEC/DOE workers were not considered in the assessment. Children were not considered in this area because they were not likely to have been visitors at the sites receiving drinking water from the Clinch River. The exposed population due to milk and meat ingestion for these two combined areas was estimated to be between 50-500 adults and 25-100 children. The estimates for fish ingestion and external exposure along the Clinch River have been previously addressed for the area between CRM 21.0 and CRM 0.0.

The routes of exposure analyzed for the Kingston Steam Plant included drinking water, fish ingestion (previously discussed), external exposure to shoreline sediments (previously discussed), and ingestion of milk or meat. The estimated number of individuals exposed due to drinking water was 300-3,000 adults (adults only, as children were not expected to be at this facility). The ingestion of milk and meat could have affected between 50-500 adults in this area and between 25-100 children (estimates include those living in the City of Kingston). The exposed population estimates for fish ingestion and exposure to shoreline sediments are the same for the entire length of the lower Clinch River considered in this analysis.

The final area of interest was the City of Kingston. Individuals living in the City of Kingston could have potentially been exposed via drinking water, fish ingestion (previously discussed), external exposure from shoreline sediments (previously discussed), and ingestion of milk or meat (included in the Kingston Steam Plant area). The exposed population estimates for drinking water are 800-4,000 adults and 200-2,000 children. The exposed adult populations could be either resident (lived and worked in the city of Kingston) or transient (lived in Kingston, but worked in another city or vice versa). As previously mentioned the estimated number of individuals for the fish ingestion pathway and for the external exposure pathway are combined for all five reaches of the lower Clinch River considered in this study. The estimates for the exposed populations via the ingestion of milk or meat were previously provided for the Kingston Steam Plant and Kingston areas.

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8.0 ESTIMATION OF RADIONUCLIDE CONCENTRATIONS IN THE FLESH OF CLINCH RIVER FISH

Consumption of fish from the Clinch River is one of the most important pathways by which off-site individuals may have been exposed to waterborne radionuclides released from the X-10 facilities. Section 8 describes an analysis of historical measurements of radionuclide concentrations in Clinch River fish and water that was performed to provide a basis for evaluation of this potential exposure pathway. The analysis concentrated on the estimation of site-specific bioconcentration factors for each of the four radionuclides of concern (^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co). A bioconcentration factor (also called concentration factor or bioaccumulation factor) is the ratio of radionuclide in fish to radionuclide in water, assuming equilibrium conditions, and is used to calculate radionuclide concentrations in fish as a function of the measured or modeled radionuclide concentration in the water. A bioconcentration factor is at best an approximation describing the net result of several processes, including uptake of radionuclide from the water, uptake through the food chain and from silt, and physiological factors affecting retention or elimination of the radionuclide by the fish.

The primary goal of this analysis was the development of subjective probability distributions describing the uncertainty about the mean bioconcentration factor for each radionuclide, for the specific conditions that existed in the Clinch River between 1944 and 1991. The available data on historic radionuclide concentrations in Clinch River fish and water were first analyzed in depth (Section 8.1). These data, together with information in the published literature and information on site-specific water conditions, were used to develop probability distributions describing site-specific bioconcentration factors for ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co (Section 8.2). Each distribution for a bioconcentration factor was summarized as a central value with minimum and maximum values describing the uncertainty about the mean bioconcentration factor for that radionuclide. Each distribution consists of alternative possible mean values of the bioconcentration factor for a sample of fish taken by a reference angler; the distribution does not describe the entire range of bioconcentration factors that might have occurred for individual fish.

In addition to the bioconcentration factors, adjustment factors were developed for use with two specific situations (Section 8.3). The first of these situations is the estimation of concentrations of radionuclides in fish caught below the mouth of White Oak Creek (e.g., by a reference individual at CRM 20.5), where fish could have been exposed to a partially mixed plume or could have spent time in White Oak Creek. The second situation is the estimation of concentrations of ^{90}Sr in whole fish. Anecdotal information indicates that some people have consumed fish patties made of ground fish, including both flesh and bones. Because ^{90}Sr, in particular, is observed in higher concentrations in whole fish than in flesh alone, an adjustment factor was developed to predict the concentrations that may have occurred in fish patties.

The annual average radionuclide concentrations in fish obtained with the bioconcentration factors are summarized in Section 8.4 and Appendix 8B. These predicted concentrations are also compared with the available measurements as a check on the validity of the bioconcentration factors.

The emphasis in this Task 4 report is on the doses and risks to reference individuals who consumed fish from the Clinch River below White Oak Creek during the period 1944-1991. For the sake of completeness, two related issues have also been included (Section 8.5): (1) fish from White Oak Lake or White Oak Creek that might have gone over White Oak Dam and been caught soon after in the Clinch River (e.g., a single "hot" fish); and (2) fish caught in the Tennessee River or Watts Bar Reservoir below the entry of the Clinch River. Many more people might have consumed fish from the Tennessee River or Watts Bar Reservoir than from the Clinch River; additionally, an individual may have consumed a larger quantity of fish from the Tennessee River-Watts Bar Reservoir system than from the Clinch River.

8.1 Description of Historical Measurements of Radionuclide Concentrations in Clinch River Fish and Water

Historical data on radionuclide concentrations in fish and water of the Clinch River are available from a series of scientific studies carried out in 1948 and in the early 1960s (Clinch River studies, Section 8.1.1), from annual reports of environmental monitoring data (1957-1991; Section 8.1.2), and from recent sampling studies performed for remediation activities (Section 8.1.3). This study has concentrated on the four radionuclides of concern: ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co. Other radionuclides reported either occasionally or regularly in fish from White Oak Lake and the Clinch River include ^{89}Sr and ^{90}Y (normally included in the ^{90}Sr value), ^{144}Ce/^{144}Pr and other rare earths, ^{32}P, ^{131}I, ^{40}K, ^{65}Zn, ^{95}Zr/Nb, ^{99}Tc, ^{110m}Ag, ^{125}Sb, ^{134}Cs, ^{212}Bi, ^{214}Bi, ^{214}Pb, ^{226}Ra, ^{228}Ac, ^{232}Th, ^{234}U, ^{235}U, ^{238}U, ^{238}Pu, and ^{239}Pu (Krumholz, 1954; USPHS, 1960; Struxness et al., 1967; ORNL, 1968; 1977; 1981; UCC, 1976; Oakes et al., 1976; TVA, 1986). In most cases, the reported values for fish represent the radionuclide concentrations in the edible tissues (flesh); the samples were cleaned and processed as if for normal human consumption. Concentrations in fish bones or whole fish were reported in a few cases. Most measurements were reported in units of activity per wet weight of fish (e.g., pCi or Bq per kg wet weight); unless otherwise specified, radionuclide concentrations in fish in this report are for edible tissues (flesh) only. Reported measurements of ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co in the flesh (edible tissues) of Clinch River fish between 1944 and 1991 are summarized in Appendix 8A for fish caught between CRM 0 and CRM 20.8 (the area of concern). Values in the tables in Appendix 8A have been converted to units of Bq kg^{-1} wet weight and (in most cases) rounded to 2 significant digits. Measurements of water concentrations were reported in units of activity per liter of water (e.g., pCi or Bq per L). Measured annual average water concentrations (in Bq L^{-1}) are summarized in Table 6.1. The limitations of the available data and the selection of data for use in this analysis are described in Sections 8.1.4 and 8.1.5, respectively.

8.1.1 Scientific Studies of Clinch River Fish

Reports from a 1948 study (Knobf, 1951) and from the Clinch River studies (USPHS, 1960; Morton, 1961; 1962; 1965; Cowser and Snyder, 1966; Nelson and Griffith, 1966) provided radionuclide concentrations in fish in terms of average concentrations from a number of individually sampled fish (Tables 8A-1, 8A-2, 8A-3, and 8A-4). Averages were provided by species (e.g., carp [*Cyprinus carpio*] or crappie [*Pomoxis* sp.]) or group of species (e.g., bottom-feeders, sight-feeders). In some cases locations

were given in terms of Clinch River miles (e.g., CRM 14.6); in other cases data were obtained over a range of locations (e.g., CRM 4.5-19.1). Sampling periods varied from a specified month (October 1948; February, May, and September 1960; May 1963) to a 2 ½-year period (January 1960 to June 1962). The February, May, and September 1960 samples are a subset of the 1960-1962 data. For crappie, average concentrations of ^{137}Cs in monthly samples were reported for May 1962 to April 1963 (Nelson and Griffith, 1966). Nelson and Griffith (1966) also reported the average ^{137}Cs concentration in Clinch River water during the same time period (0.029 Bq L^{-1} at CRM 14.4).

The data for fish collected in October 1948 were provided in terms of counts per minute per gram (Knobf, 1951). These activity levels were converted to units of pCi kg^{-1} (later to Bq kg^{-1}) based on the results of calculations (including corrections for geometry and self-absorption) performed on fish collected during the same study from White Oak Creek and White Oak Lake. The reported sample means for 1948 are not true sample means because an effort was made in the original study to select fish with higher levels of radioactivity for detailed analysis. Although that effort was not entirely successful, the actual means of these samples are expected to be lower than the values reported. Analyses of the fish from White Oak Creek and White Oak Lake showed that 96.8% of the activity in the flesh was due to ^{137}Cs and 3.2% due to ^{90}Sr. Essentially all activity in the bone was due to ^{90}Sr.

8.1.2 Annual Reports of Environmental Monitoring Data

Annual monitoring reports provided measured radionuclide concentrations in fish from 1965 to 1991 (ORNL, 1968; 1969; 1973; 1977; 1978; 1979; 1980; 1981; 1982; 1985; UCC, 1972; 1973; 1974; 1975; 1976; 1983; MMES, 1984; 1986; 1987; 1988b; 1989b; 1990b; 1991b; 1992; many of these data as well as data from the Clinch River studies are summarized in Oakes et al., 1982). Data were reported for most but not all years in this period (Tables 8A-5 and 8A-6). Concentrations were provided by species of fish (usually 1 to 5 species) and usually but not always by sampling location. Sampling reportedly took place during the spring or summer, but actual sampling dates were not given. In most cases, reported concentrations are the measurements of one or more composite samples of 10 fish (range, 6-20 fish; arithmetic averages of composite samples were reported in the case of multiple composite samples). For 1985, reported concentrations were taken from a graph in the absence of a table, except for a few values that were given in the text. For 1988, concentrations in bluegill (*Lepomis macrochirus*) reported as ash weight were converted to fresh weight using an approximate conversion factor (80 g fresh weight per g ash weight), based on data for 1991 which provided both ash weight and fresh weight concentrations for bluegill.

Measured concentrations of radionuclides in water were included in the reports for the years 1957 to 1990 at CRM 4.5, CRM 14.5, or both (ORNL, 1958; 1959; 1960; 1961; 1962; 1963; 1964; 1965; 1966; 1967; 1968; 1969; 1970; 1971; 1973; 1977; 1978; 1979; 1980; 1981; 1982; 1985; UCC, 1972; 1973; 1974; 1975; 1976; 1983; MMES, 1984; 1986; 1987; 1988a; 1989a; 1990a; 1991a). At CRM 4.5, the reported annual average concentrations were the averages of quarterly composites of daily grab samples. At CRM 14.5, the reported annual average concentrations were the averages of quarterly composites from

a continuous proportional sampler. For most years, a concentration measurement was reported for one location or the other; for a few years, concentrations were reported at both locations. These data and their uncertainties are discussed in Sections 6.1-6.3; the reported concentrations are summarized in Table 6.1.

8.1.3 Recent Sampling Studies

Recent compilations of fish data (e.g., the Clinch River Remedial Investigation reports, e.g., ORNL, 1992; and the TVA In Stream Contaminant Study, e.g., TVA, 1986) were also examined. The primary focus in these recent sampling studies was on chemical contaminants such as mercury and PCBs, rather than on radionuclides. These summaries provided insufficient detail on the dates and locations of sampling, and in general did not offer much additional information for this specific study.

8.1.4 Limitations of the Environmental Data

The historical measurements of radionuclide concentrations in fish from the Clinch River are valuable in that they are reported measurements for fish that were actually caught in the Clinch River during the time period of importance. The project team gave serious thought to using the data directly in the dose assessment. However, the data sets are subject to several important limitations.

First, no data at all are available for Clinch River fish from 1944-1947 and 1949-1959. These periods are expected to include some of the highest radionuclide concentrations in water (and hence in fish) for the whole project period. Secondly, no effort was made in many of the studies to perform systematic sampling of fish with respect to species or location along the river. Neither the set of species sampled nor the set of sampling locations was consistent across years. The largest set of samples (1960-1962), perhaps the most representative in terms of species sampled, was reported only in summary form. Limited information is available in terms of a breakdown by species, and no information was included in terms of a breakdown of the data by sampling date or location. Most of the fish appear to have been caught in relatively few sampling efforts in late spring or early summer. The May 1963 data are also not available by sampling location. The data that are available for later years (1978-1990) suggest that differences in radionuclide concentrations among species for any given sampling location, or differences in concentrations with sampling location for any one species, are neither large nor consistent (Figure 8.1). Concentrations in fish caught between CRM 10-15 are usually but not always greater than for fish caught between CRM 2-5. Fish sampled at or not far below CRM 20.8; generally (but again not always) had much higher radionuclide concentrations than fish sampled below CRM 15. Fish near CRM 20.8 could swim in or near the plume coming from White Oak Creek or into and out of White Oak Creek Embayment (see Section 8.3.1), and thus could have been exposed to greater or more varied concentrations than fish caught below CRM15.

In addition, little information is available on the seasonal fluctuations in radionuclide concentrations in Clinch River fish. Most of the fish samples were obtained during the spring or summer, and thus the measurements might not be representative of concentrations throughout the year. Of the data sets that covered a longer time period, all but one provided only the mean concentrations for the entire time period, with no

information on monthly or quarterly values. Monthly concentrations of ^{137}Cs in white crappie (*Pomoxis annularis*) for a 12-month period (May 1962-April 1963) showed considerable variation, but no evidence of significant seasonal fluctuations (Figure 8.2; Nelson and Griffith, 1966). However, without information on monthly variations in the ^{137}Cs concentration in the water, it is not possible to determine whether the observed variation is due to fluctuations in the water concentration or to seasonal variation in uptake of ^{137}Cs by the fish. It is also not possible to know whether significant fluctuations may have occurred in other fish species or with other radionuclides.

Sample size was a problem with many of the reported measurements. The Clinch River studies in general had adequate sample sizes, even though information on the individual fish samples (dates and locations of sampling, concentrations in individual fish) could not be located. However, many of the reported environmental monitoring measurements were based on a small number of composite samples (usually only 1) of 10 fish for each species. In other words, 10 fish caught at one time point were combined and measured once, to give the reported concentrations of radionuclides in fish for the year. For 1965-68 and 1971-74, concentrations were reported for only 2 samples per year, 1 for each of 2 edible fish species, with unspecified locations (gizzard shad [*Dorosoma cepedianum*] is not considered an edible species and was not included in this study). For 1975-77, the locations were provided, but at most 2 samples were available for locations below CRM 15. For 1975, the report specifically claimed that the "data contain some anomalies which may be due to the limited number of samples collected and the sensitivities of the methods of analysis used" (UCC, 1976). For 1986-1991, data were limited to 1 species and 1 location below CRM 15. No data for Clinch River fish were located for 1964 or 1969-70.

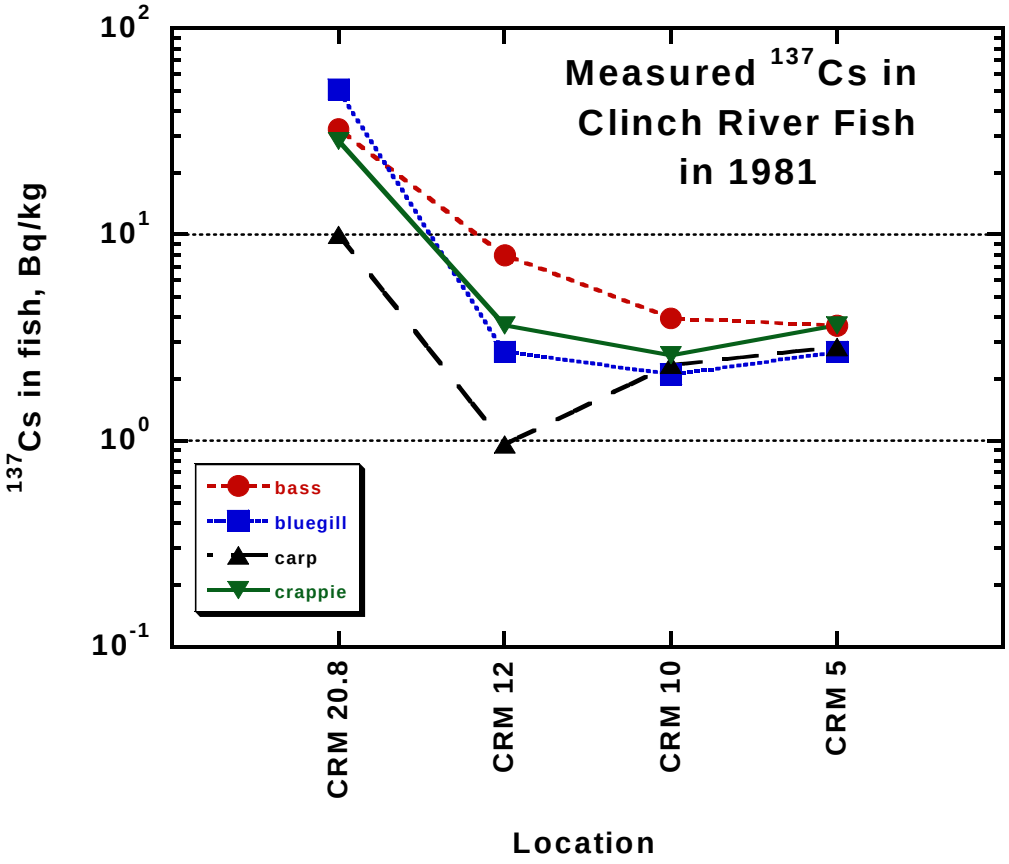

Figure 8.1 Measured concentrations of ^{137}Cs in Clinch River fish in 1981 (ORNL, 1982). The points represent reported mean concentrations in composite samples for the species and sampling locations indicated.

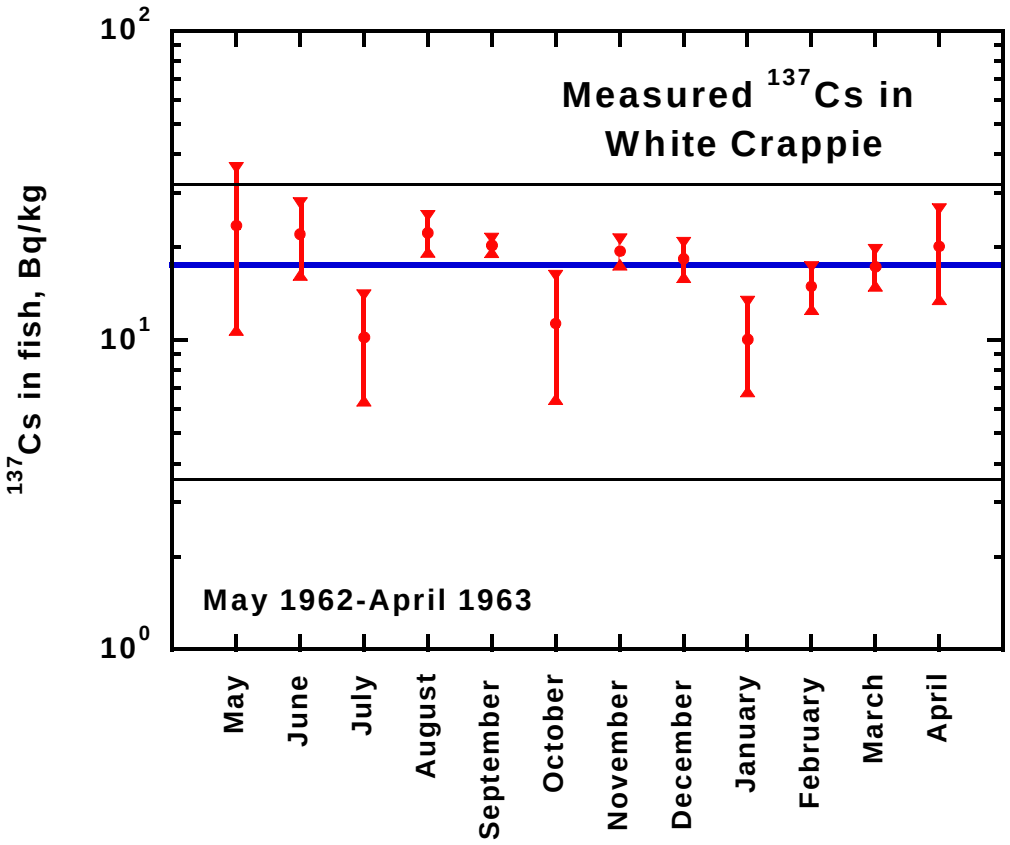

Figure 8.2 Monthly average concentration of ^{137}Cs in white crappie (Nelson and Griffith,1966). The dark circles indicate the mean measured concentration for the month indicated. The vertical lines indicate 1 standard error (1 SE) about the mean value. The thick horizontal line indicates the mean value for the entire period; the standard error about the mean is indicated by the thin horizontal lines.

Incomplete processing of samples could have contributed to biased measurements of ^{90}Sr in some samples throughout the period of interest. Although the fish samples were reportedly prepared as if for human consumption--removal of scales, head, guts, and bones--it is possible that many samples did contain some bone fragments. Because ^{90}Sr is concentrated in the bones, it is possible that some values reported for ^{90}Sr may be high, due to the presence of even very small bone fragments.

Analytical difficulties contribute an additional source of uncertainty to all of the measurements. In particular, measured concentrations below about 0.04 Bq per kg fish or L water (1 pCi per kg or L) are subject to large counting errors unless large amounts of sample were used (I. L. Larsen, Oak Ridge National Laboratory, personal communication). In many of the environmental monitoring reports, obtaining a low value for the estimated percentage of the Maximum Permissible Intake (MPI; an intake of fish comparable to a daily intake of 2.2 L of water containing the Maximum Permissible Concentration of the radionuclide for a period of one year; Oakes et al., 1982) appears to have been of more concern than reporting an accurate concentration--if the value was less than 1% of the MPI, the exact value of the concentration in fish was not important. Thus many of the fish concentrations in the environmental monitoring reports may be off by a factor of 3 (300%) or more. The reported measurements for ^{106}Ru are considered especially uncertain, particularly for cases involving limited samples and low concentrations of ^{106}Ru in the water, due in part to the small gamma peak usually measured to indicate the presence of that isotope.

8.1.5 Selection of Data Sets for Use in the Analysis

Because no fish data at all are available for a number of years of concern (1944-1947, 1949-1959, 1964, 1969-70), and because most of the data sets are incomplete with respect to fish species, sampling dates, and sampling locations, the project team decided that use of estimated bioconcentration factors (expressed as a probability distribution describing the uncertainty about the mean bioconcentration factor for a given radionuclide) for all years would provide the most appropriate estimates of the mean radionuclide concentrations in Clinch River fish that would be relevant to anglers who harvested fish at regular intervals. Selected data sets were used as part of the basis for development of the bioconcentration factors (Section 8.2), but due to their limitations (Section 8.1.4) could not be used as the sole basis for development of bioconcentration factors.

Based on the discussions in Sections 8.1.1 to 8.1.4, the following data sets were selected for use in the development of site-specific bioconcentration factors (Sect. 8.2) or specified adjustment factors (Sect. 8.3):

Scientific studies of Clinch River fish:

- January 1960-June 1962 (including individual subsets from February, May, and September 1960): ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co

- May 1962-April 1963: ^{137}Cs in white crappie

C May 1963: ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co

Environmental monitoring reports:

C 1978-1985: ^{137}Cs, ^{90}Sr, and ^{60}Co

Each of these data sets included at least two edible species of fish and identified the corresponding sampling locations; in addition, measured annual average radionuclide concentrations in Clinch River water were available for these years or sampling periods. To avoid possible biases, no values (for fish or water) reported as being below the levels of detection ("less-than" values) were included in the analyses of the bioconcentration factors or the adjustment factors (for the years listed above, such values comprised a very small fraction of the data). The 1975 data, which were reported to be anomalous (UCC, 1976), also were not used.

Additional data sets that included only one species of fish between CRM 0 and CRM 15, or for which no measured water concentrations were available, were used as additional information in comparison of measured and modeled concentrations in fish (Section 8.4). These included fish data for 1948, 1976-1977, and 1986-1991.

Edible species of fish in this study consisted primarily of largemouth bass (*Micropterus salmoides*), bluegill (*Lepomis macrochirus*), carp (*Cyprinus carpio*), crappie (*Pomoxis* sp.), and buffalo (*Ictiobus* sp.). Additional species included suckers (*Catostomus* sp.), redhorse (*Moxostoma* sp.), carpsuckers (*Carpiodes carpio*), quillback (*Carpiodes cyprinus*), channel catfish (*Ictalurus punctatus*), flathead catfish (*Pylodictis olivaris*), white bass (*Roccus chrysops*), smallmouth bass (*Micropterus dolomieu*), sauger (*Stizostedion canadense*), and drum (*Aplodinotus grunniens*). "Bottom-feeders" in the May and September 1960 samples included carp, suckers, buffalo, redhorse, carpsuckers, quillback, channel catfish, and flathead catfish (Morton, 1962). "Sight-feeders" in the 1960-1962 samples included white crappie (*Pomoxis annularis*), bluegill, white bass, largemouth bass, sauger, drum, and catfish (primarily channel catfish; Cowser and Snyder, 1966).

Observed site-specific bioconcentration factors were calculated from data on fish caught between CRM 2 and CRM 5 and measured or adjusted water concentrations at CRM 4.5 or from data on fish caught between CRM 10 and CRM 15 and measured or adjusted water concentrations at CRM 14.5 (Section 8.2). (If measured water concentrations were available at only one location, they were adjusted to estimate concentrations at the other; see Section 6.3.) Fish caught above CRM 18 were not used for estimation of bioconcentration factors due to the likelihood that they might have spent time in White Oak Creek Embayment, the plume from White Oak Creek, or areas above the confluence of White Oak Creek, any of which could have been misleading in terms of estimating a bioconcentration factor. However, data for fish caught at CRM 20.8 were used to estimate an adjustment factor for prediction of radionuclide concentrations in fish at the CRM 20.5 receptor location (Section 8.3.1). Comparisons of concentrations in whole fish vs. fish flesh were based on the 1960-1962 data set (Section 8.3.2).

8.2 Estimation of Site-Specific Bioconcentration Factors

Site-specific bioconcentration factors (concentration in fish per unit concentration in water, Bq kg^{-1} per Bq L^{-1}, or L kg^{-1}) were developed from three sets of information: observed bioconcentration factors calculated from measured concentrations in fish and water, published information on the behavior of the radionuclides in rivers and in fish, and information on local water conditions. The bioconcentration factors were described in terms of probability distributions; each distribution for a bioconcentration factor was summarized as a central value with minimum and maximum values describing the uncertainty about the mean bioconcentration factor for that radionuclide. Each distribution consists of alternative possible mean values of the bioconcentration factor for a sample of fish taken by a reference angler; the distribution does not describe the entire range of bioconcentration factors that might have occurred for individual fish. The probability distribution developed for each radionuclide is described in the following sections.

The bioconcentration factors were not derived solely from the measured concentrations in fish and water, due to the limitations of the available data (Section 8.1.4). In particular, most of the fish samples were obtained during the spring or summer, while the water data accounted for the whole calendar year. Thus events affecting the water concentrations in the last half of a year would not be reflected in the fish samples for that year, but in some cases might affect the fish concentrations for the following year (depending on the radionuclide and the biological half-life in fish). In addition, the species sampled and the number of samples taken were not consistent throughout the data sets, nor was the information available to calculate true (properly weighted) mean concentrations in fish for each year. Thus, while the measurements are extremely valuable for indicating or confirming the magnitude of the bioconcentration factor for any given radionuclide, they are insufficient in themselves for calculation of truly representative values.

The observed bioconcentration factors were obtained by dividing an average radionuclide concentration in fish flesh by the average water concentration for that location and time period (Table 8.1; values in the table are rounded to 2 significant digits). Note that the values for fish are means of the sample means listed in Appendix 8A; values were not weighted for the number or mass of fish in each sample, as that information was not available in most cases. The fish data for 1960-1962 and 1963 included fish sampled between CRM 4.5 and CRM 19.1; water concentrations for CRM 14.5 were used with these data to calculate the corresponding bioconcentration factor. For 1960-1962, the annual average water concentrations for the three years were averaged to obtain an estimated average for the whole period. It should be noted that some of the fish sampled in 1960 were reported separately, in addition to being included in the 1960-1962 totals; calculations were made with both the total set and the 1960 subsets. For the May 1962-April 1963 fish sample, an average concentration of ^{137}Cs in water at CRM 14.4 was reported for the same time period (Nelson and Griffith, 1966); this value was used rather than annual averages for 1962 or 1963. Known uncertainties in the fish and water measurements were not propagated through the calculations of the observed bioconcentration factors but were considered in developing the subjective probability distributions for the uncertain mean value.

Table 8.1 Summary of observed bioconcentration factors (BCFs), based on reported measurements of radionuclides in Clinch River fish and water.

	CRM 2-5[a]			CRM 10-15[b]		
	Average Concentrations		Observed	Average Concentrations		Observed
Radionuclide and Year	**Fish ($Bq\ kg^{-1}$)**	**Water ($Bq\ L^{-1}$)**	**BCF ($L\ kg^{-1}$)**	**Fish ($Bq\ kg^{-1}$)**	**Water ($Bq\ L^{-1}$)**	**BCF ($L\ kg^{-1}$)**
^{137}Cs						
1960	18	0.085	210	30	0.094	320
1960-62				27[c]	0.048	560
1963				17[c]	0.094	180
May 1962-April 63				18	0.029	600
1978	8.7	0.026	330	3.2	0.043	73
1979	3.2	0.00074	4300	26	0.00074	36,000
1980	3.1	0.0027	1100	4.4	0.0030	1500
1981	3.2	0.0039	830	3.3	0.0043	770
1982	2.1	0.0023	94	3.5	0.025	140
1983	2.9	0.0094	310	4.8	0.010	470
1985	2.0	0.0057	350	2.3	0.0063	370
^{90}Sr						
1960	24	0.35	68	29	0.39	75
1960-62				14[c]	0.23	58
1963				1.9[c]	0.12	16
1978	0.15	0.0037	39	0.19	0.0041	46
1979	0.17	0.012	14	0.57	0.015	38
1980	0.38	0.025	15	0.36	0.028	13
1981	0.34	0.048	7.0	0.30	0.053	5.6
1982	0.41	0.061	6.8	0.33	0.067	4.9

Table 8.1 (Continued)

Radionuclide and Year	CRM 2-5[a] Average Concentrations Fish (Bq kg^{-1})	CRM 2-5[a] Average Concentrations Water (Bq L^{-1})	CRM 2-5[a] Observed BCF (L kg^{-1})	CRM 10-15[b] Average Concentrations Fish (Bq kg^{-1})	CRM 10-15[b] Average Concentrations Water (Bq L^{-1})	CRM 10-15[b] Observed BCF (L kg^{-1})
^{90}Sr (continued)						
1983	0.58	0.067	8.6	0.39	0.074	5.3
1984	0.24	0.037	6.6	0.58	0.041	14
1985	0.54	0.064	8.5	0.43	0.070	6.1
^{106}Ru						
1960	9.9	14	0.73	3.9	15	0.26
1960-62				4.8[c]	11	0.44
^{60}Co						
1960	2.9	0.20	15	6.2	0.22	29
1960-62				2.6[c]	0.17	16
1978	0.18	0.0041	43	0.31	0.0059	52
1979	0.16	0.0015	100	0.53	0.0019	280
1980	0.10	0.0071	14	0.18	0.0078	23
1981	0.14	0.0032	44	0.11	0.0035	32
1982	0.040	0.020	2.0	0.10	0.022	4.5
1985	0.11	0.0047	23	0.16	0.0052	30

[a] Fish were caught between CRM 2 and CRM 5 (see Appendix 8A for details); water concentrations were reported for CRM 4.5 or adjusted for CRM 4.5 based on reported measurements at CRM 14.5 (see Sections 6.1, 6.3).
[b] Fish were caught between CRM 10 and CRM 15 (see Appendix 8A for details); water concentrations were reported for CRM 14.5 or adjusted for CRM 14.5 based on reported measurements at CRM 4.5 (see Sections 6.1, 6.3).
[c] Fish were caught between CRM 4.5 and CRM 19.1.

8.2.1 Cesium-137

For ^{137}Cs in the Clinch River, the observed bioconcentration factors calculated from measured fish and water ranged from 73 to 36,000 (Table 8.1); all but 4 of the values were between 200 and 2,000. Values calculated from the ambient potassium concentrations ranged from 420-3300 (1961) or 660-2300 (1989-90), depending on the trophic level of the fish. Calculations were based on Vanderploeg et al. (1975), who described the bioconcentration factor for cesium in turbid water (suspended solids >50 ppm) as 3,000/[K] for piscivorous fish and 1,000/[K] for nonpiscivorous fish, where [K] = the concentration of potassium in the water (ppm). Reported values of [K] in the Clinch River range from 0.9-2.4 ppm (CRM 5.5, 1961; Morton, 1962) and 1.3-1.5 ppm (CRM 6, 1989-90; USDOE, 1996b), and suspended solids have typically been above 50 ppm (Struxness et al., 1967). These data resulted in estimated bioconcentration factors from 420-1,100 for nonpiscivorous fish and from 1,200-3,300 for piscivorous fish.

Table 8.1 includes an historical study of the bioconcentration of ^{137}Cs by Clinch River fish. Nelson and Griffith (1966) described a study of white crappie (*Pomoxis annularis*) collected monthly between May 1962 and April 1963 at about CRM 10. In all, 112 individual fish were sampled, or about 10 fish per month (Table 8A-4; Figure 8.2). The average ^{137}Cs concentration in the edible flesh of the fish was about 18 Bq kg^{-1} (reported as 1.059 disintegrations per minute per gram). The average concentration of ^{137}Cs in water at CRM 14.4 during the same time period was 0.029 Bq L^{-1} (reported as 1.767×10^{-3} disintegrations per minute per gram). Together, these measurements give a bioconcentration factor of 600 for ^{137}Cs in fish.

Reported values of bioconcentration factors for ^{137}Cs vary with the type of aquatic environment (e.g., eutrophic or mesotrophic), the trophic level of the fish, the turbidity of the water, and the concentration of potassium in the water (Vanderploeg et al., 1975; Blaylock, 1982). Recommended default values range from 2000 if water conditions are unknown (IAEA, 1994) to 5000/[K] and 15,000/[K] for nonpiscivorous and piscivorous fish, respectively (Poston and Klopfer, 1988). In a review of published bioconcentration factors, Blaylock (1982) reported a range of 281 to 4850 for the mean bioconcentration factor for ^{137}Cs in freshwater fish. A bioconcentration factor of 700 was obtained for bluegill in White Oak Lake (Kolehmainen and Nelson, 1969; Kohlemainen, 1972), but that value included the entire fish (except stomach contents), rather than just the flesh.

Based on this information, the mean bioconcentration factor for ^{137}Cs in Clinch River fish (all edible species) was described by a log-triangular subjective probability distribution with a central value of 600 and minimum and maximum values of 120 and 3000, respectively. The central value was selected on the basis of the detailed study by Nelson and Griffith (1966). The minimum and maximum values represent an uncertainty factor of 5. This subjective probability distribution accounts for the observed bioconcentration factors, the ranges predicted from observed concentrations of potassium, inclusion of all trophic levels of fish, local water conditions (e.g., less eutrophic than White Oak Lake), and the potential changes in turbidity after the construction of Melton Hill Dam. Comparisons of measured ^{137}Cs concentrations in the

flesh of Clinch River fish (by location and species) with estimated mean concentrations (all edible species) obtained with this bioconcentration factor are shown in Section 8.4.

8.2.2 Strontium-90

For ^{90}Sr in the Clinch River, the observed bioconcentration factors calculated from measured fish and water ranged from 4.9 to 75 (Table 8.1). Values calculated from the ambient calcium concentrations ranged from 1.2 to 5.3. Calculations were based on Vanderploeg et al., 1975, who described the bioconcentration factor for strontium as exp (5.18 - 1.21 ln [Ca]), where [Ca] = the concentration of calcium in water (ppm). Reported values of [Ca] in the Clinch River range from 18-63 ppm (Morton, 1962; MMES, 1986; 1992; USDOE, 1996a).

One historical study was made of the bioconcentration of stable strontium by Clinch River fish. For white crappie (*Pomoxis annularis*) collected in 1962-63 (see Section 8.2.2 above), Nelson (1966) reported a bioconcentration factor for stable strontium of 1.0 for fish flesh. Based on measurements of specific activity, the corresponding bioconcentration factor for ^{90}Sr was expected to be similar. The value of 1.0 is lower than any of the empirical factors obtained for ^{90}Sr from other data sets in the present study; this may be attributable to more careful separation of bones and flesh in Nelson's research efforts than in the other studies.

Strontium acts as a chemical homologue of calcium, and the concentration of stable strontium in fish is inversely related to the calcium concentration in water (Ophel and Judd, 1969). In fish, strontium is rapidly concentrated in the bones (Nelson and Griffith, 1966). Observed values of the bioconcentration factor for ^{90}Sr vary with both the trophic level of the fish and the calcium concentration in the water (Blaylock, 1982). Poston and Klopfer (1988) recommend a bioconcentration factor of 50 for ^{90}Sr in fish flesh when the calcium concentration in the water is not known. A bioconcentration factor of 300 has been recommended for whole fish (Pally and Foulquier, 1979). Blaylock (1982) has reported a range of published mean values from about 1 to 200 for edible tissues (flesh) of freshwater fish.

Based on this information, the mean bioconcentration factor for ^{90}Sr in Clinch River fish (all edible species) was described by a log-triangular subjective probability distribution with a central value of 10 and minimum and maximum values of 1 and 100, respectively. The minimum and maximum values represent an uncertainty factor of 10. This subjective probability distribution accounts for the observed bioconcentration factors, the ranges predicted from observed concentrations of calcium, inclusion of various amounts of bones with flesh samples (both in analyses and in individual human diets), and local water conditions. Comparisons of measured concentrations of ^{90}Sr in the flesh of Clinch River fish (by location and species with estimated mean concentrations (all edible species) obtained with this bioconcentration factor are shown in Section 8.4.

8.2.3 Ruthenium-106

Observed bioconcentration factors for ^{106}Ru in the Clinch River were based on a very small set of samples. Values based on measured fish and water concentrations ranged from 0.26 to 0.73 (Table 8.1).

Ruthenium in freshwater environments is known to form both organic and inorganic complexes which are not biologically active, and therefore ruthenium does not accumulate to any extent in fish tissue (Poston and Klopfer, 1988). Blaylock (1982) reported bioconcentration factors for ^{106}Ru in Clinch River fish from 0.1 to 3.5 for fish flesh and 1.8 for whole fish. Factors of 100 or greater have been recommended for less eutrophic freshwater systems (Poston and Klopfer, 1988), while a value of 10 was recommended by Thompson et al. (1972).

Based on this information, the mean bioconcentration factor for ^{106}Ru in Clinch River fish (all edible species) was described by a log-triangular subjective probability distribution with a central value of 1 and minimum and maximum values of 0.1 and 10, respectively. The minimum and maximum values represent an uncertainty factor of 10. This subjective probability distribution accounts for the observed bioconcentration factors, the small amount of data available for ^{106}Ru in Clinch River fish, and currently very incomplete knowledge about the behavior of ruthenium in freshwater fish. Comparisons of measured concentrations of ^{106}Ru in the flesh of Clinch River fish (by location and species) with estimated mean concentrations (all edible species) obtained with this bioconcentration factor are shown in Section 8.4.

8.2.4 Cobalt-60

Observed bioconcentration factors for ^{60}Co in the Clinch River calculated from measured fish and water concentrations ranged from 2.0 to 280 (Table 8.1); all but 3 values are between 10 and 150. Cobalt is an essential element for fish, with the highest concentrations found in the kidney and spleen (Poston and Klopfer, 1988). In general, the higher the organic content of a freshwater system, the lower the bioconcentration factor in the fish. Recommended generic bioconcentration factors for ^{60}Co have ranged from 20 (Thompson et al., 1972) to 330 (Poston and Klopfer, 1988). Poston and Klopfer (1988) recommend a value of 30 for eutrophic systems and 330 for mesotrophic or unclassified systems (based on Vanderploeg et al., 1975). Blaylock (1982) reported values from 5 to about 50 for fish in eutrophic environments and from about 230 to 600 for mesotrophic environments. The Clinch River is roughly midway between eutrophic and mesotrophic. Values also depend on the trophic level of the fish.

Based on this information, the mean bioconcentration factor for ^{60}Co in Clinch River fish (all edible species) was described by a log-triangular subjective probability distribution with a central value of 30 and minimum and maximum values of 6 and 150, respectively. The minimum and maximum values represent an uncertainty factor of 5. This subjective probability distribution accounts for the observed bioconcentration factors, local water conditions (between eutrophic and mesotrophic), and inclusion of all trophic levels of fish. Comparisons of measured concentrations of ^{60}Co in the flesh of Clinch River fish (by location and

species) with estimated mean concentrations (all edible species) obtained with this bioconcentration factor are shown in Section 8.4.

8.2.5 Summary of Subjective Probability Distributions for Bioconcentration Factors

The mean bioconcentration factors for all four radionuclides are summarized in Table 8.2. Each bioconcentration factor is described as a log-triangular subjective probability distribution with a central value and minimum and maximum values describing the uncertainty about the mean bioconcentration factor for that radionuclide. Each distribution consists of alternative possible values of the mean bioconcentration factor for a sample of fish taken by a reference angler; the distribution does not describe the entire range of bioconcentration factors that might have occurred for individual fish.

Table 8.2 Subjective probability distributions for mean bioconcentration factors (L kg^{-1}) for radionuclides in the flesh of Clinch River fish.

Radionuclide	Central Value	Minimum Value	Maximum Value	Distribution Shape
^{137}Cs	600	120	3000	log-triangular
^{90}Sr	10	1	100	log-triangular
^{106}Ru	1	0.1	10	log-triangular
^{60}Co	30	6	150	log-triangular

The subjective distributions for bioconcentration factors that are described here represent a combination of empirical and theoretical approaches for estimating the mean radionuclide concentrations in fish potentially harvested by a reference individual, given a specified annual average concentration in the river water. The ranges of the distributions are intended to be wide enough to account for uncertainties in the observed bioconcentration factors, the representativeness of the available data, and local water conditions, as well as for inclusion of all trophic levels of fish and the lack of full knowledge about the behavior of radionuclides in freshwater systems. As a check on the validity of these bioconcentration factors, the predicted annual average radionuclide concentrations in fish were compared with available measurements (Section 8.4).

8.3 Adjustment Factors for Specific Situations

This section describes adjustment factors for use with two specific situations. The first of these is the estimation of concentrations of radionuclides in fish caught below the mouth of White Oak Creek (e.g., by a reference individual at CRM 20.5), where fish could have been exposed to a partially mixed plume or could have moved in and out of White Oak Creek. The second situation is the estimation of concentrations of ^{90}Sr in fish patties made from ground whole fish.

8.3.1 Radionuclide Concentrations in Fish Near the Mouth of White Oak Creek

The location where a fisherman is likely to encounter the highest concentrations of radionuclides in fish is at CRM 20.8, near the confluence of White Oak Creek with the Clinch River. A comparison of fish concentrations measured at or near CRM 20.8 with concentrations measured in fish caught further downstream (CRM 10-15) demonstrates a substantial difference, with CRM 20.8 concentrations being larger by about a factor of 5 to 7 on the average. The differences in any given set of samples range from a factor of near 1 to about 50. Thus radionuclide concentrations in fish at CRM 20.8 can be expected to be significantly higher than at downstream locations, but the exact difference is highly uncertain. The higher fish concentrations may be due to fish that have migrated out of the White Oak Creek embayment or that have spent a considerable amount of time in the incompletely mixed plume of water discharged into the Clinch River from White Oak Creek.

An adjustment factor derived from ratios of measured fish at CRM 20.8 to measured fish at CRM 10-15 was used to estimate the radionuclide concentrations in fish (all edible species) at CRM 20.5 (one of the receptor locations identified in Section 7) from the calculated concentrations at CRM 14. The adjustment factor was described as a subjective log-triangular distribution with a central value of 6 and minimum and maximum values of 1 and 30, respectively. The large range reflects the high uncertainty involved in estimating the annual average concentration of radionuclides in fish at CRM 20.5. Comparisons of measured radionuclides in the flesh of Clinch River fish (by species) caught between CRM 18 and CRM 20.8 with estimated mean concentrations at CRM 20.5 are shown in Section 8.4.

8.3.2 Consumption of Fish Patties

Anecdotal information indicates that some people have consumed fish patties made of ground fish, including both flesh and bones (ORNL, 1985). Consumption of fish bones is a concern primarily for ^{90}Sr, because concentrations of ^{90}Sr in the bones may exceed those in the flesh by two orders of magnitude or more. Higher concentrations of ^{106}Ru in whole fish than in flesh alone were reported for fish and bones (up to a factor of 2; Cowser and Snyder, 1966) or for whole fish including the viscera (up to a factor of 15; Morton, 1962); however, fish patties would normally have included only the flesh and the bones (ORNL, 1985). An adjustment factor was developed to predict the concentrations of ^{90}Sr that may have occurred in fish patties due to inclusion of the bones.

For the fish sampled between January 1960 and June 1962 (Cowser and Snyder, 1966), radionuclide concentrations were reported for both fish flesh and whole fish. For ^{90}Sr, the whole fish to flesh ratio ranged from 1.7 to 10 (Table 8.3), with a geometric mean of 3.9. From values for bioconcentration factors in whole fish reported by Blaylock (1982), a whole fish-to-flesh ratio of 980 was obtained for ^{90}Sr; this value was based on a lower bioconcentration factor in fish flesh than was obtained in the present study. The actual situation is expected to depend greatly on how the fish are cleaned and processed before consumption. Based on the data from Cowser and Snyder (1966), the adjustment factor for ^{90}Sr was described as a subjective log-triangular distribution with a central value of 4 and minimum and maximum

values of 1 and 24, respectively. Support for this selected range was obtained from dose estimates prepared for consumption of fish flesh or of carp patties (ORNL, 1985). Dose estimates were based on all radionuclides measured (^{137}Cs, ^{90}Sr, ^{60}Co, ^{238}Pu, ^{239}Pu, ^{234}U, ^{235}U, and ^{238}U). Per kg consumed, the estimated average doses (effective dose equivalents) were about 4 to 13 times higher for carp patties than for carp flesh (ORNL, 1985).

Table 8.3 Comparison of ^{90}Sr concentrations in flesh and total fish (data from Cowser and Snyder, 1966)[a].

Fish Species	^{90}Sr Concentrations (Bq kg^{-1})		Total-to-Flesh Ratio
	Flesh	Total fish[b]	
Carp	18.5	190	10
Carpsucker	20	35	1.7
Buffalo	8.9	31	3.5

[a] Fish samples were obtained between CRM 4.5 and CRM 19.1, from January 1960 to June 1962.

[b] Includes flesh and bones.

8.4 Summary of Predicted Radionuclide Concentrations in Clinch River Fish

The predicted radionuclide concentrations in Clinch River fish, in terms of 95% subjective confidence intervals (2.5th and 97.5th percentiles) and central values (50th percentiles), are tabulated in Appendix 8B for the four locations of concern (CRM 0, 3.5, 14, and 20.5). These predicted concentrations were obtained using the measured or modeled concentrations in water (Section 6); the bioconcentration factors summarized in Table 8.2; and, for CRM 20.5, the adjustment factor described in Section 8.3.1. Note that water concentrations before 1960 are modeled, while for 1960 and later years, water concentrations were based primarily on measurements (to the extent available; Section 6). The 95% subjective confidence intervals on the predicted radionuclide concentrations in fish reflect the uncertainties in the water concentrations, the bioconcentration factors, and (for CRM 20.5) the adjustment factors. Each subjective distribution tabulated in Appendix 8B consists of alternative possible mean values of the annual average concentration of radionuclide in fish for a sample of fish taken by a reference angler over the course of that year; the distributions do not describe the entire range of concentrations that might have occurred for individual fish.

The predicted concentrations in fish at CRM 3.5, 14, and 20.5 were compared with available measurements at CRM 2-5, 10-14.6, and 18-20.8, respectively (Figures 8.3-8.14). Comparisons were not made for predictions at CRM 0 because no data are available below CRM 2. Data for 1960-1962 and 1963 are shown only in the figures for CRM 10-14.6 but actually represent fish sampled between CRM 4.5 and 19.1. The figures show the 95% subjective confidence interval and central value of the predictions, but only the mean values (or measurements of composite samples) for the measurements (see Section 8.1 and Appendix 8A for more details on the measured values).

As described in Section 8.1.5, data for 1948, 1976-1977, and 1986-1991 were not used in the development of the bioconcentration factors but are included in these figures. The bioconcentration factors were based in part on observed bioconcentration factors, which were calculated from means of sample means (Section 8.2); Figures 8.3-8.14 show the actual sample means by species of fish.

Of the nearly 500 measurements shown in Figures 8.3-8.14, more than three-fourths fall inside or on the edge of the 95% subjective confidence interval of the predictions, indicating that the bioconcentration factors developed in Section 8.2 do provide a realistic comparison of the relative concentrations of radionuclides in fish and water for the Clinch River. Perfect agreement between predictions and measurements is not expected in this situation for a number of reasons:

(1) The subjective probability distributions developed for bioconcentration factors (Section 2) are intended to describe the intervals in which the true values of the mean bioconcentration factors are expected to lie. Similarly, the subjective probability distribution developed for the adjustment factor for fish at CRM 20.5 (Section 8.3.1) is intended to describe the interval in which the true value of the mean adjustment factor is expected to lie. The subjective confidence intervals on the radionuclide concentrations in fish predicted with these bioconcentration factors and the adjustment factor are intended to contain the annual mean values in fish caught throughout the year from a combination of edible species; the confidence intervals are not expected to include all individual values of every fish (or sample of fish) that was caught. Thus, if a truly representative data set were available for a given year and location, the mean value of that data set would be expected to fall within the predicted range for the mean, but many individual values probably would not.

(2) A bioconcentration factor by definition is based on the assumption of an equilibrium situation between the concentration of a radionuclide in fish and the concentration in the water in which the fish is found. However, the Clinch River situation in question here is definitely not at equilibrium: some species of fish migrate up and down the river, with some species traveling farther than others (Martin et al., 1964); the amounts of radionuclides discharged to the river from White Oak Creek were not constant; the flow rates were not constant; the sampling dates for the fish are not representative of the entire year in most cases; and various species and sizes (i.e., ages) of fish and various sampling locations were not systematically represented in the sampling data. Thus the bioconcentration factors are, at best, approximations for the complex relationships between the concentrations of radionuclides in fish and the concentrations in water.

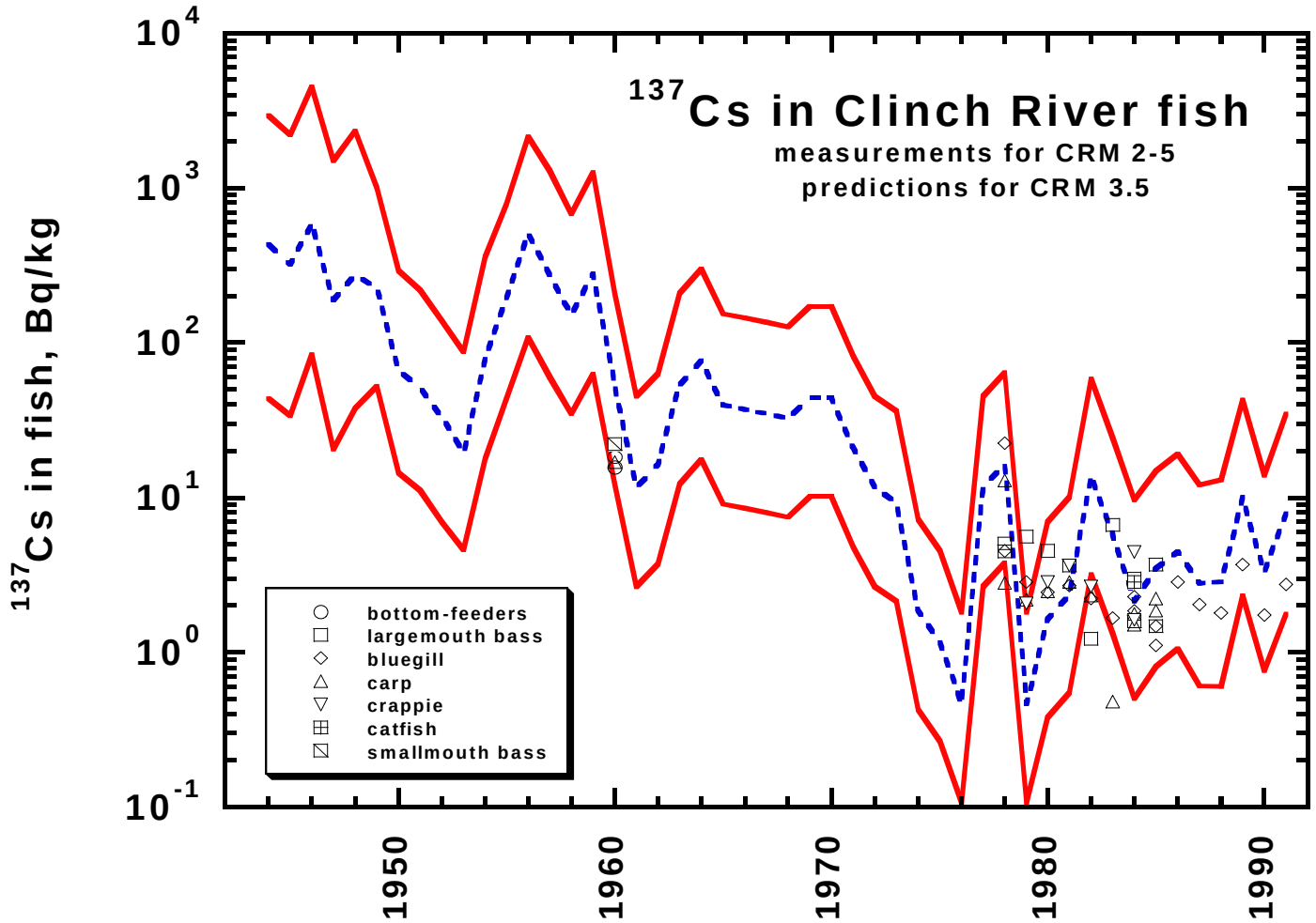

Figure 8.3. Comparison of predicted concentrations of ^{137}Cs in the flesh of edible fish at CRM 3.5 with measurements of ^{137}Cs in the flesh of fish caught between CRM 2 and CRM 5. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated.

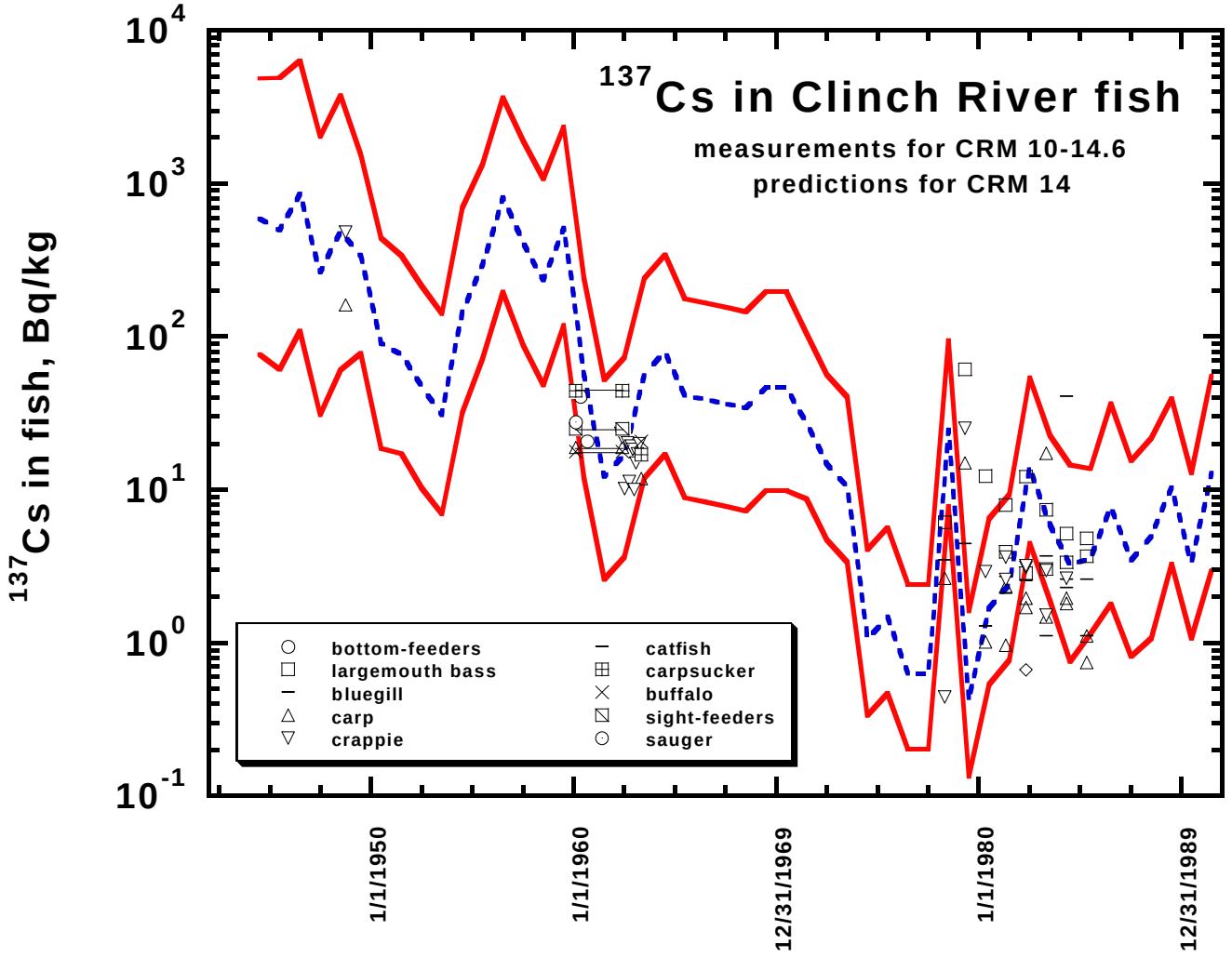

Figure 8.4. Comparison of predicted concentrations of ^{137}Cs in the flesh of edible fish at CRM 14 with measurements of ^{137}Cs in the flesh of fish caught between CRM 10 and CRM 14.6. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the species and sampling years indicated. Small horizontal lines indicate samples collected between Jan. 1960 and June 1962; these samples and the May 1963 samples were taken between CRM 4.5 and CRM 19.1. [In this graph, data are shown by month of sampling (or May if the actual date is not known); predicted annual averages are shown for July.]

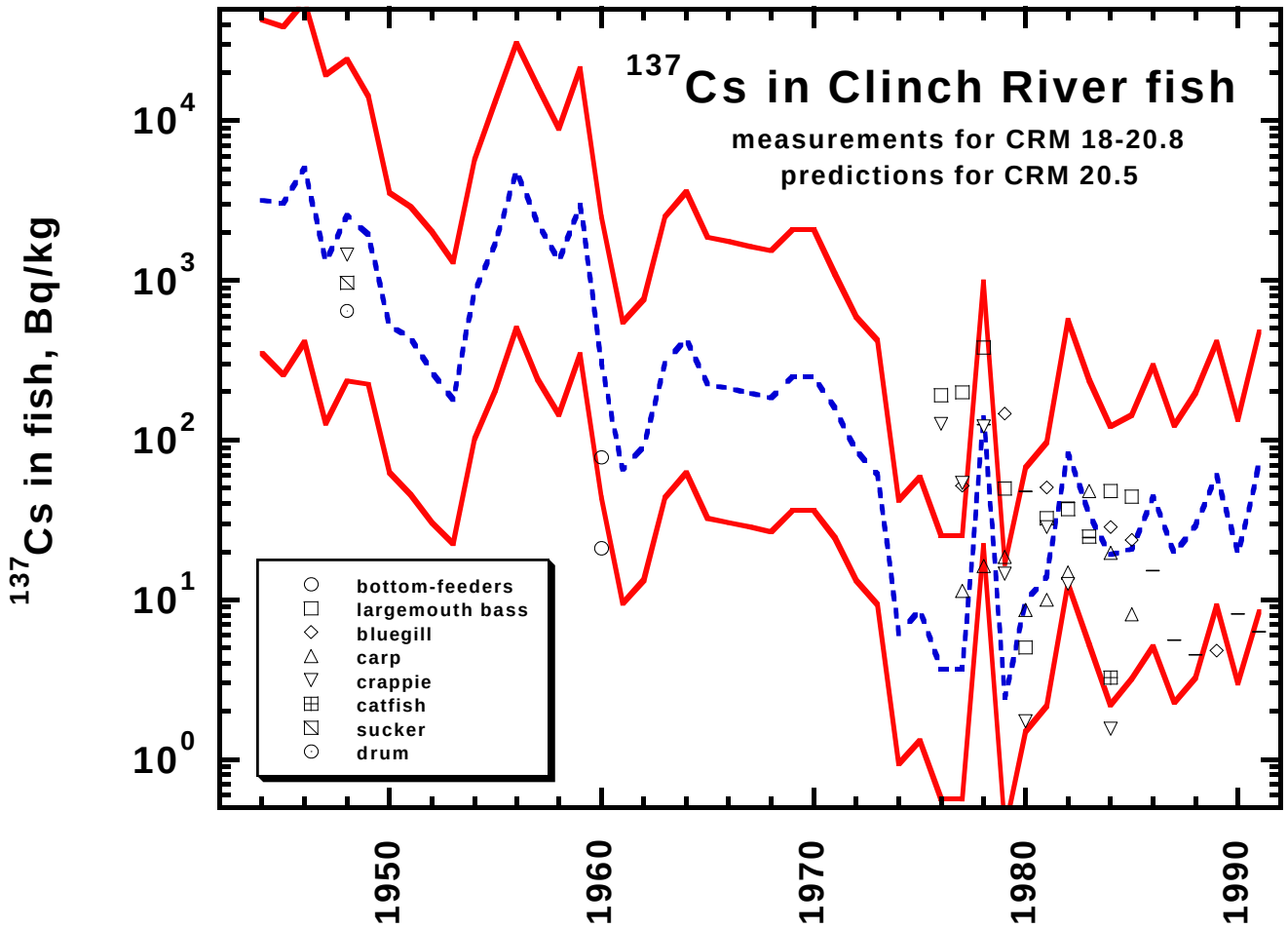

Figure 8.5. Comparison of predicted concentrations of ^{137}Cs in the flesh of edible fish at CRM 20.5 with measurements of ^{137}Cs in the flesh of fish caught between CRM 18 and CRM 20.8. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated.

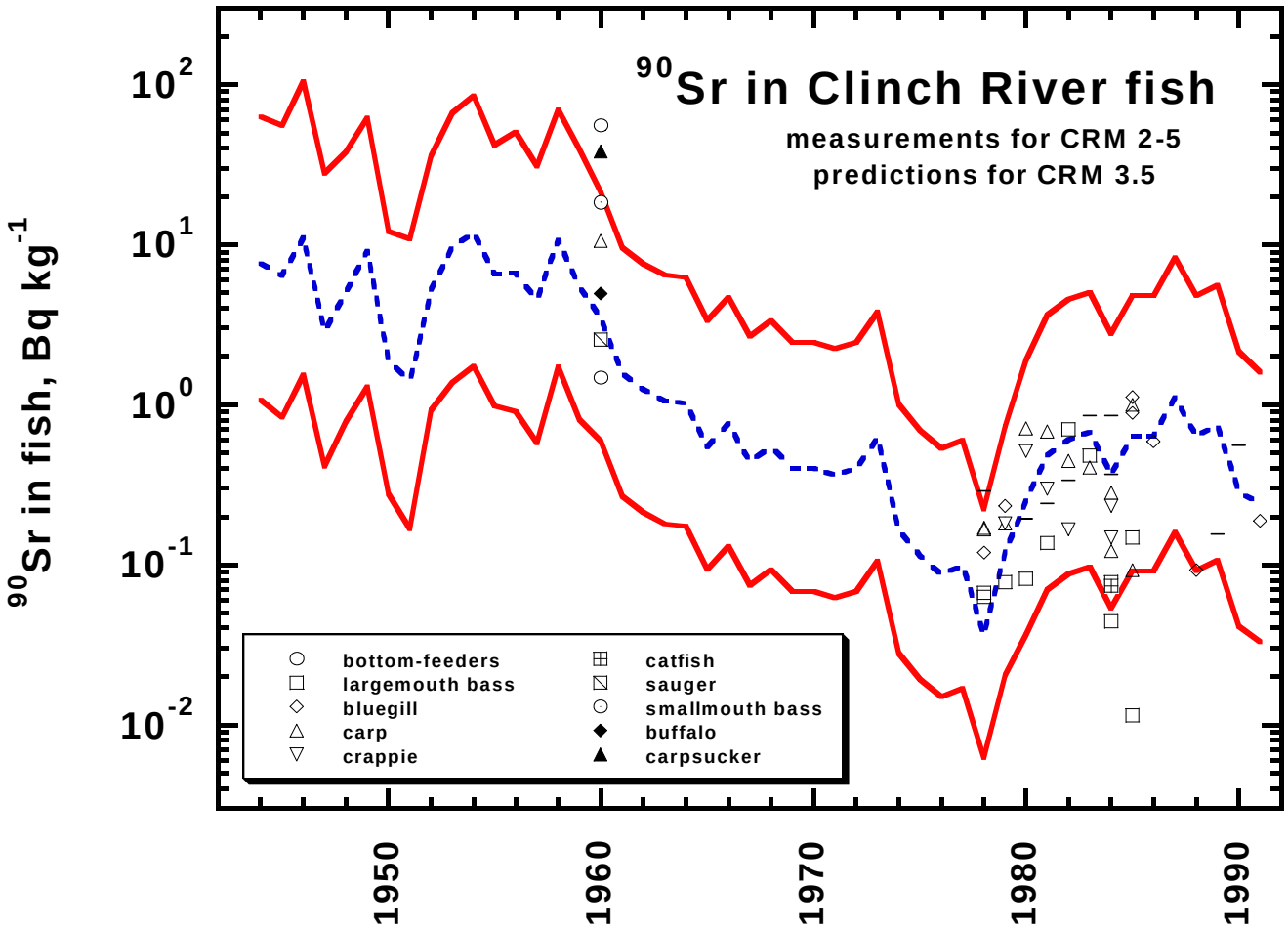

Figure 8.6. Comparison of predicted concentrations of ^{90}Sr in the flesh of edible fish at CRM 3.5 with measurements of ^{90}Sr in the flesh of fish caught between CRM 2 and CRM 5. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated.

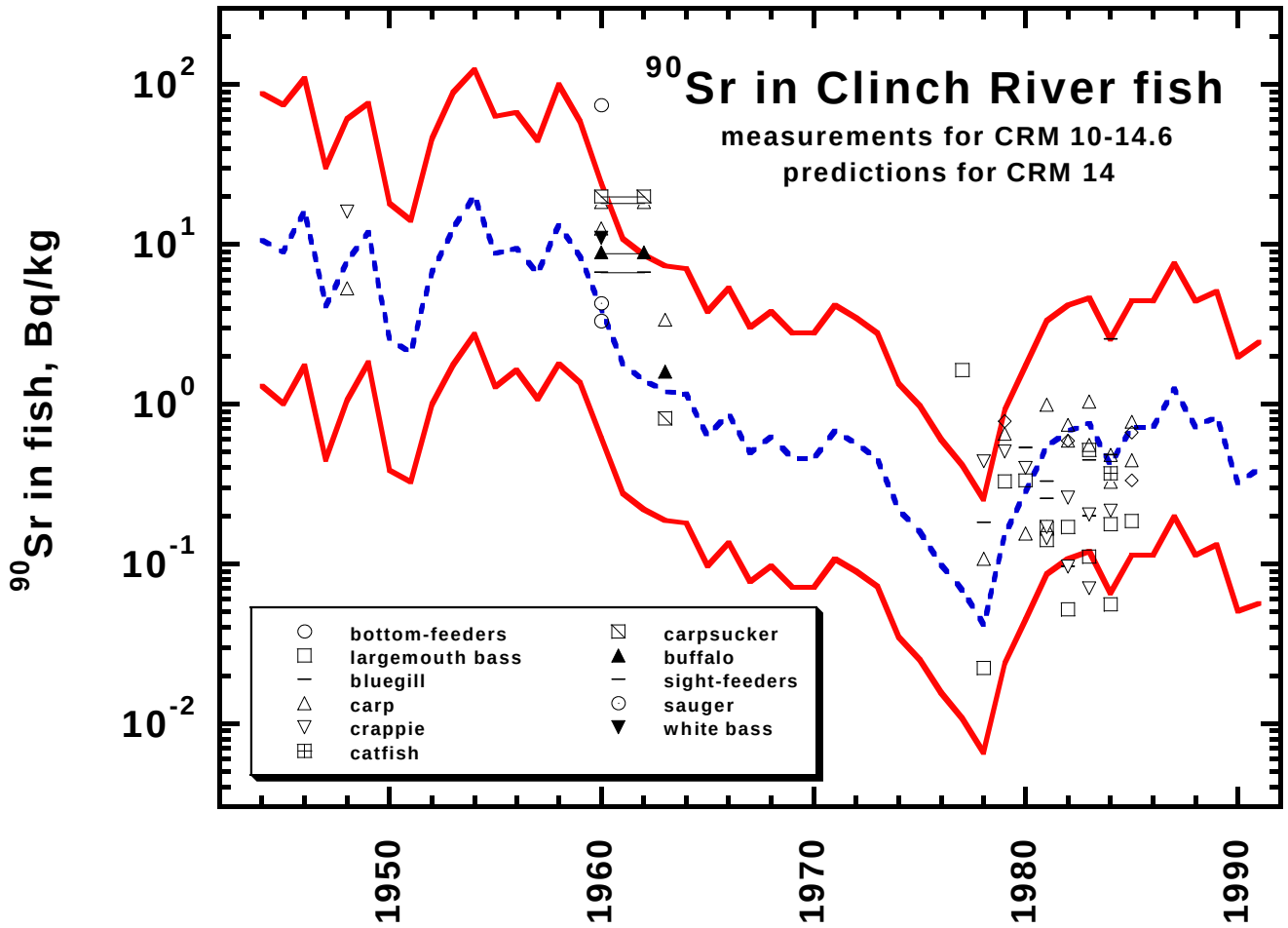

Figure 8.7. Comparison of predicted concentrations of ^{90}Sr in the flesh of edible fish at CRM 14 with measurements of ^{90}Sr in the flesh of fish caught between CRM 10 and CRM 14.6. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated. Small horizontal lines indicate samples collected between January 1960 and June 1962; these samples and the 1963 samples were taken between CRM 4.5 and CRM 19.1.

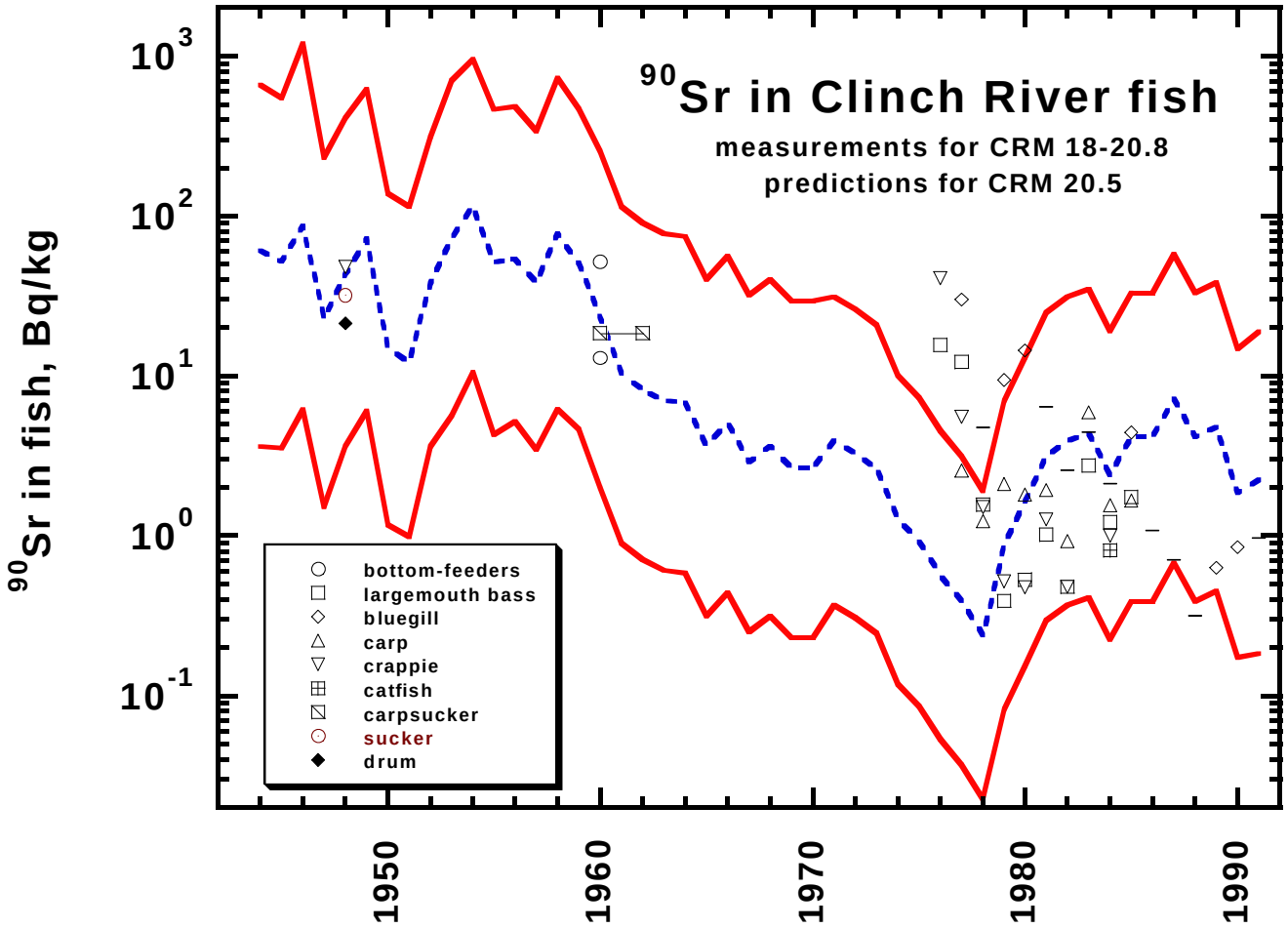

Figure 8.8. Comparison of predicted concentrations of ^{90}Sr in the flesh of edible fish at CRM 20.5 with measurements of ^{90}Sr in the flesh of fish caught between CRM 18 and CRM 20.8. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated. The small horizontal line indicates a composite sample of 4 fish collected some time between January 1960 and June 1962 at CRM 19.6.

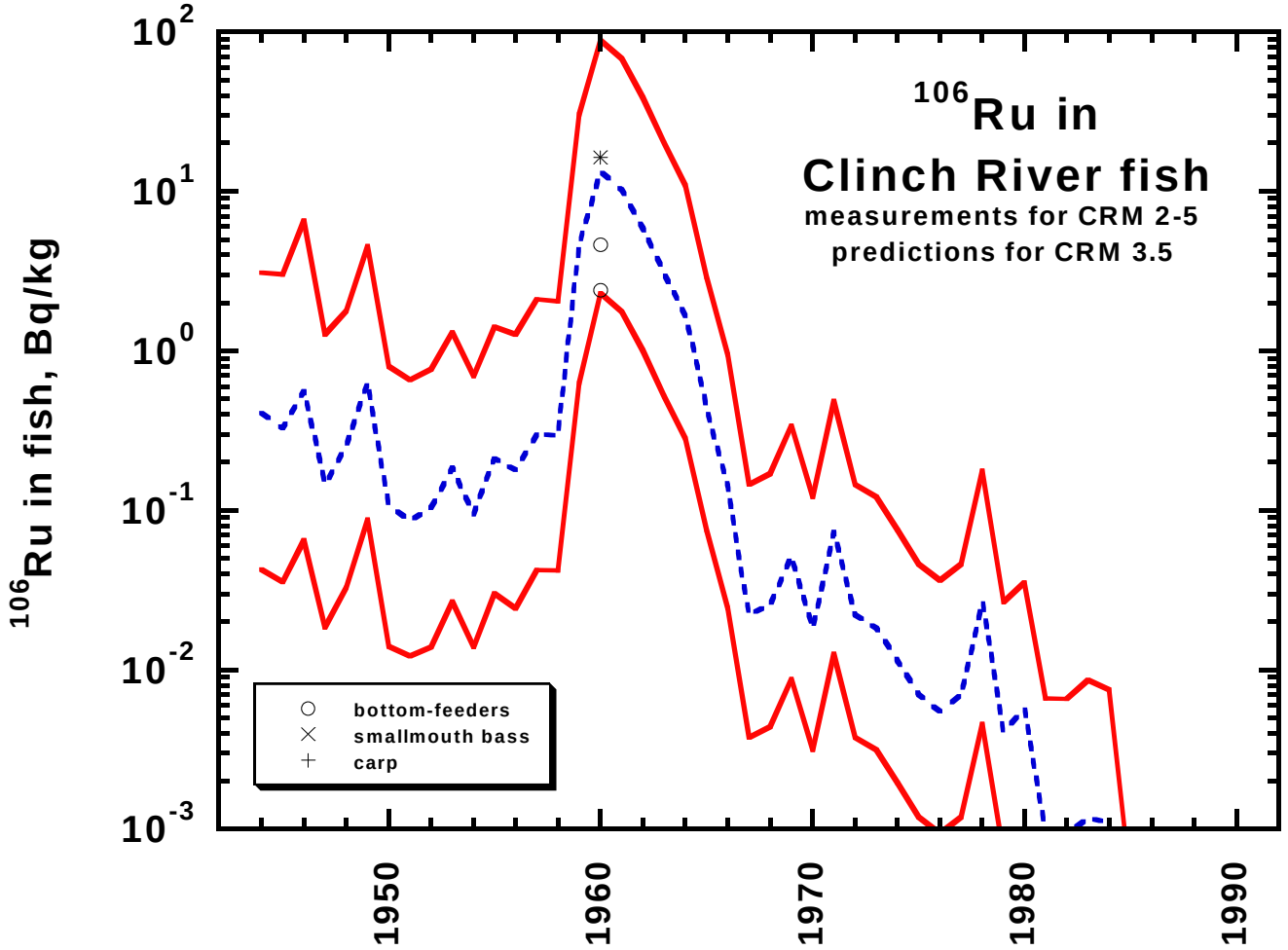

Figure 8.9. Comparison of predicted concentrations of ^{106}Ru in the flesh of edible fish at CRM 3.5 with measurements of ^{106}Ru in the flesh of fish caught between CRM 2 and CRM 5. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means for the fish species and sampling year indicated.

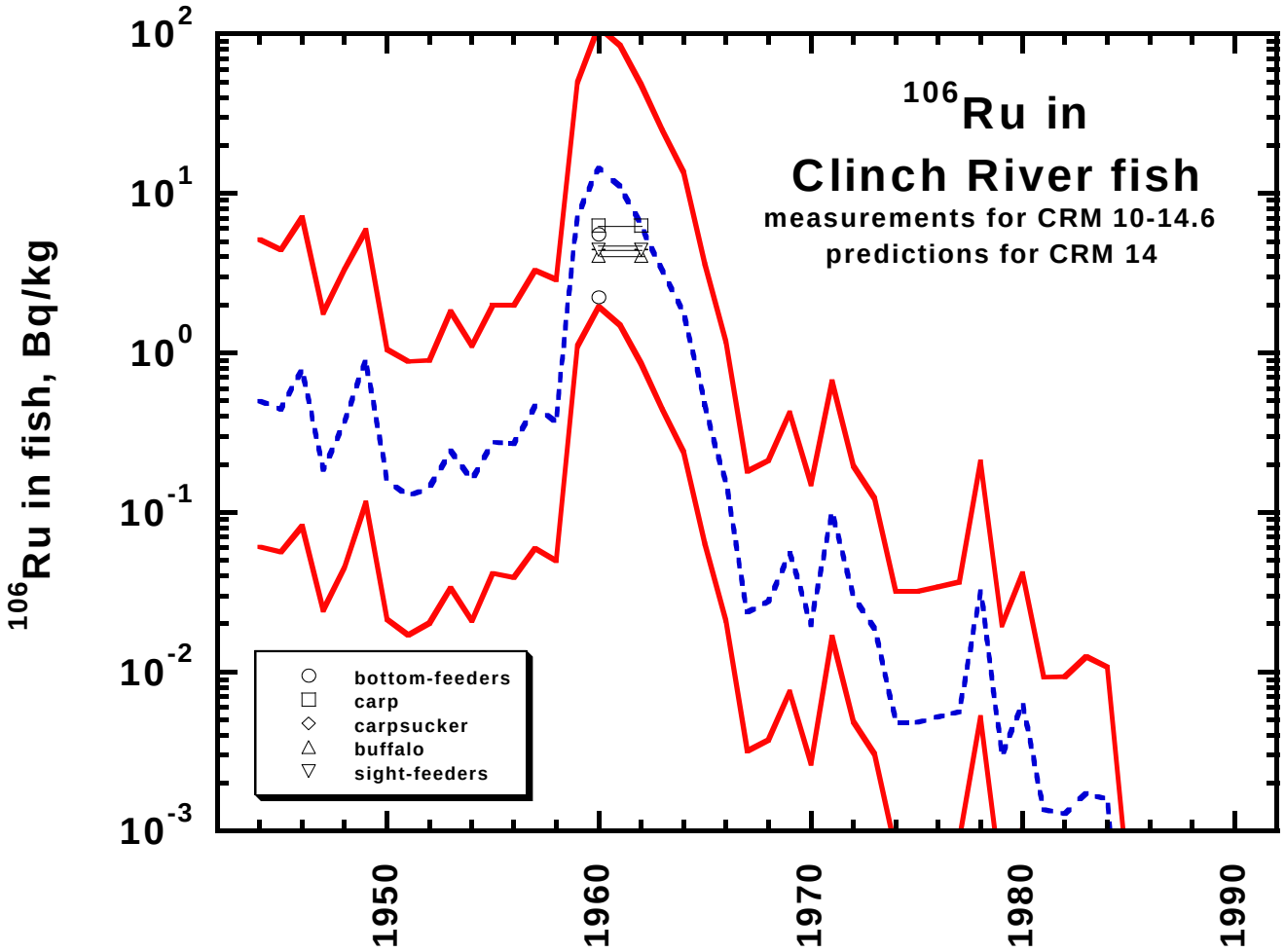

Figure 8.10 Comparison of predicted concentrations of ^{106}Ru in the flesh of edible fish at CRM 14 with measurements of ^{106}Ru in the flesh of fish caught between CRM 10 and CRM 14.6. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means for the fish species and sampling years indicated. Small horizontal lines indicate samples collected between January 1960 and June 1962; these samples were taken between CRM 4.5 and CRM 19.1.

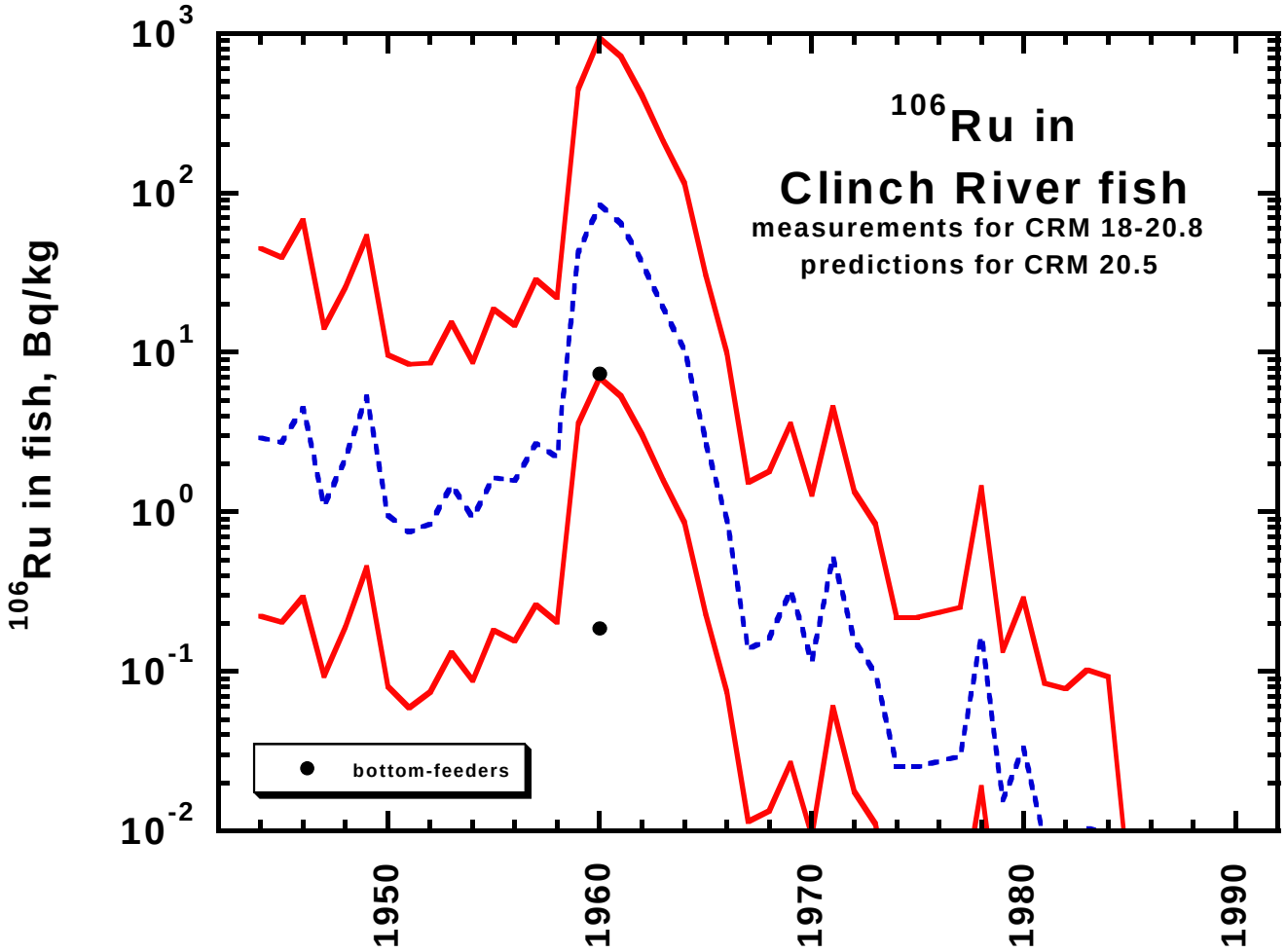

Figure 8.11. Comparison of predicted concentrations of ^{106}Ru in the flesh of edible fish at CRM 20.5 with measurements of ^{106}Ru in the flesh of fish caught between CRM 18 and CRM 20.8. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means for the fish species and sampling year indicated.

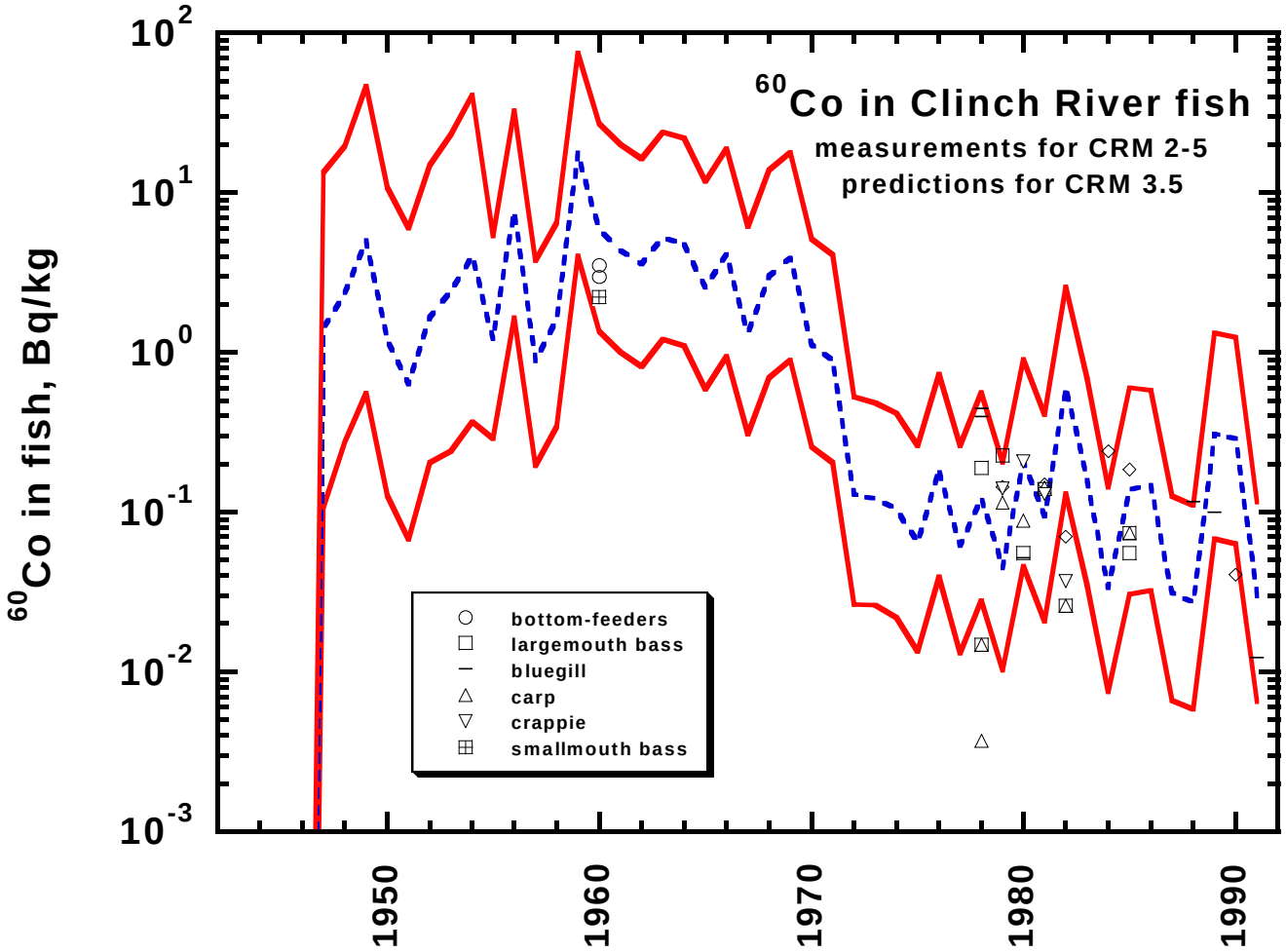

Figure 8.12. Comparison of predicted concentrations of ^{60}Co in the flesh of edible fish at CRM 3.5 with measurements of ^{60}Co in the flesh of fish caught between CRM 2 and CRM 5. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated.

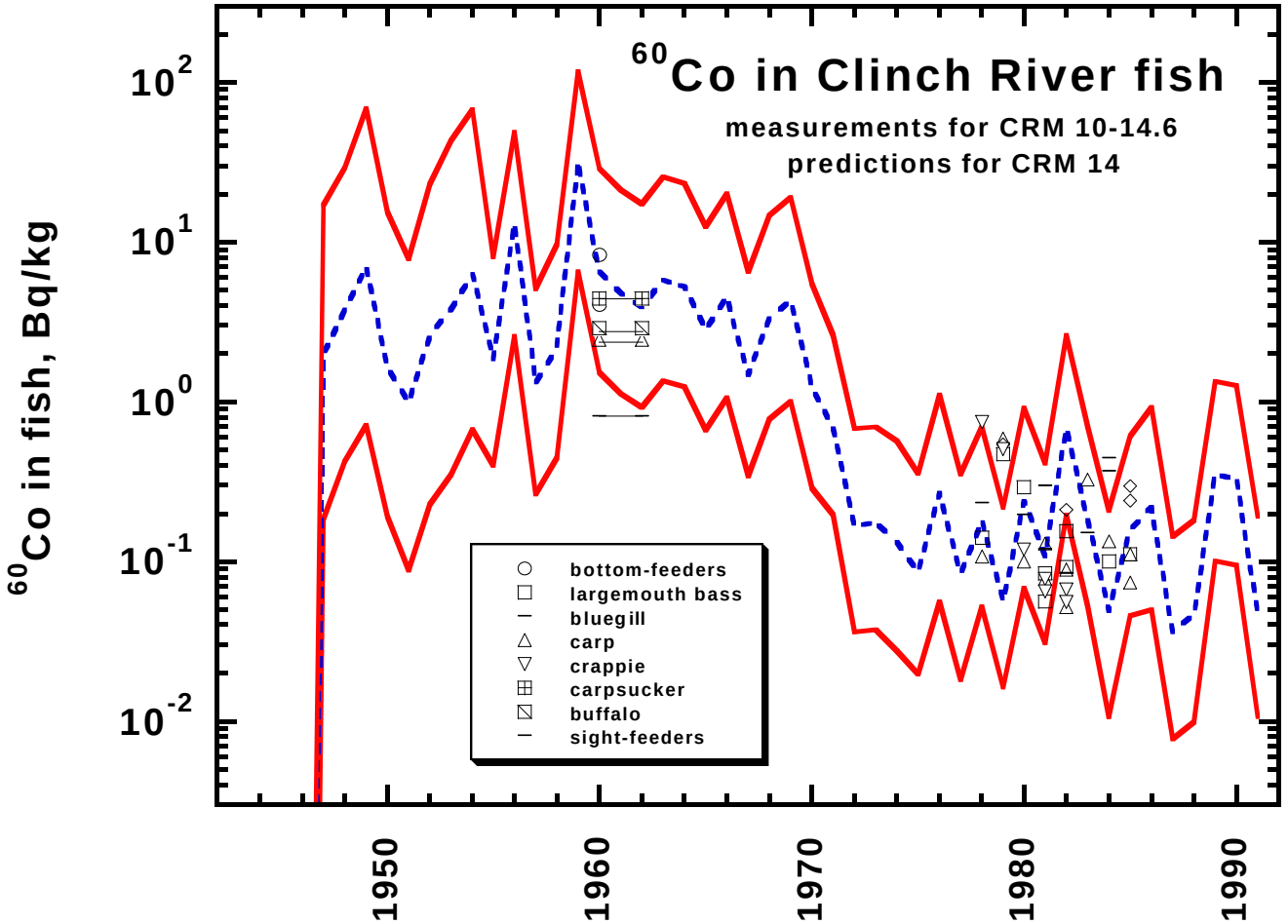

Figure 8.13. Comparison of predicted concentrations of ^{60}Co in the flesh of edible fish at CRM 14 with measurements of ^{60}Co in the flesh of fish caught between CRM 10 and CRM 14.6. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated. Small horizontal lines indicate samples collected between January 1960 and June 1962; these samples were taken between CRM 4.5 and CRM 19.1.

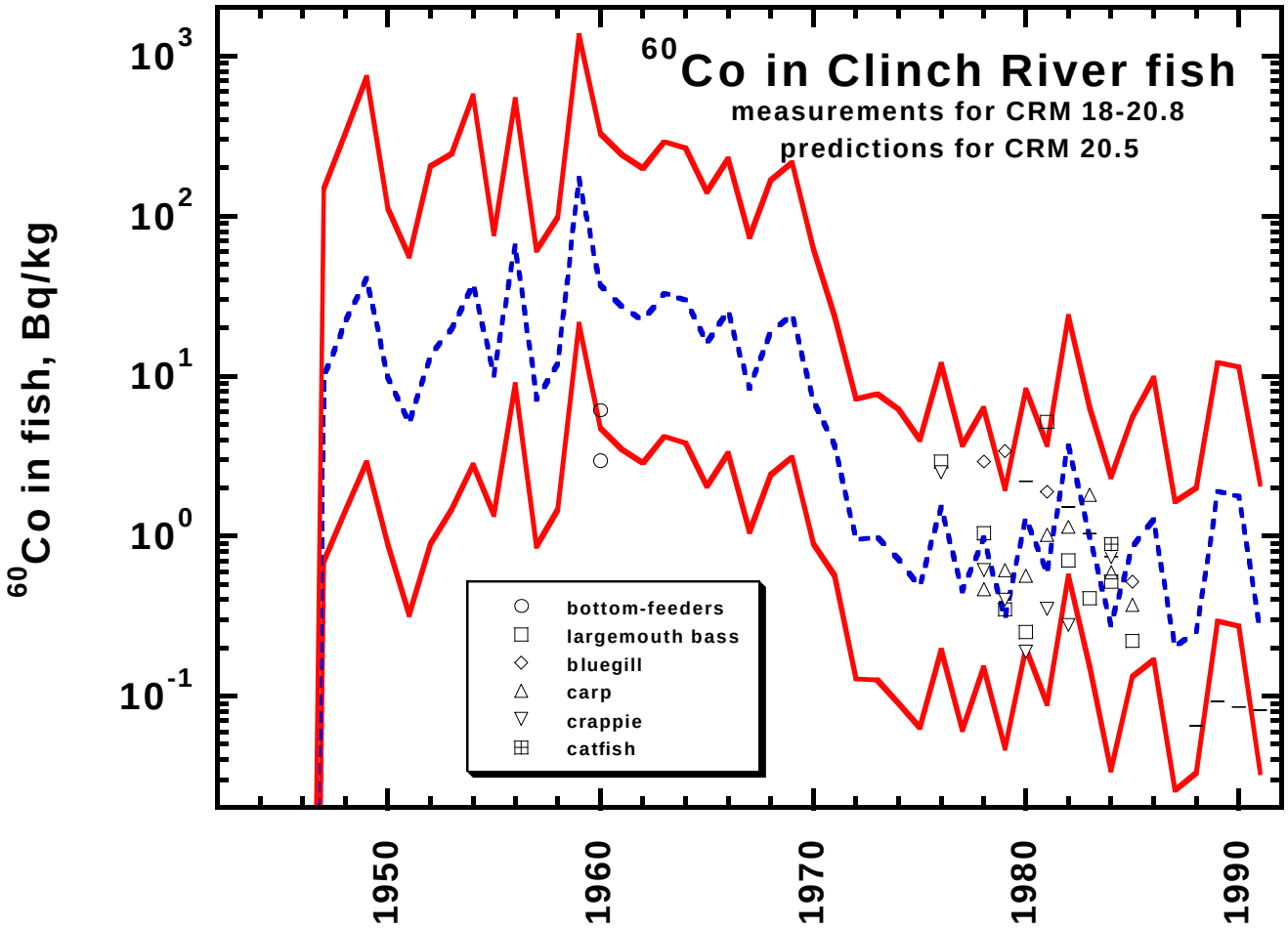

Figure 8.14. Comparison of predicted concentrations of ^{60}Co in the flesh of edible fish at CRM 20.5 with measurements of ^{60}Co in the flesh of fish caught between CRM 18 and CRM 20.8. Solid lines indicate the 95% subjective confidence interval (2.5th and 97.5th percentiles) on the predicted concentrations; the dashed line indicates the central value (50th percentile). Symbols indicate sample means or measurements of composite samples for the fish species and sampling years indicated.

(3) The graphs show predicted annual average concentrations compared with measurements taken (in most cases) at a single point in time. As described in Section 8.2, most fish samples were taken in the spring or summer, while the water concentrations are averages for the whole year. The "out-of-phase" appearance of the measured and predicted concentrations of ^{137}Cs and ^{60}Co in 1978-1979 and 1982 are probably due to events (e.g., increased discharges) that occurred in the last half of a year, affecting the water concentrations for that year but the fish concentration for the following year, depending on the biological half-life of the radionuclide in fish. The 1960-1962 samples (shown with short horizontal lines) cover a 2 ½-year period, but the fish were probably not sampled uniformly across that time period. In particular, the value for carpsuckers shown for ^{90}Sr at CRM 18-20.8 represents a composite sample of 4 fish, presumably taken at one time, but when during that period is not known.

(4) The measurements shown in the graphs are not directly comparable to the predicted values in terms of location. For example, the graphs for CRM 20.5 include measurements from fish taken at CRM 18-19 (1948), CRM 19.6 (1960; 1960-62 for ^{90}Sr), CRM 20 (1976), and CRM 20.8 (1977-1991). Fish reported for CRM 20.8 could include fish actually taken from White Oak Creek, from upstream of CRM 20.8, or from the other side of the river where the plume was not mixed. The graphs for CRM 14 include measurements from fish taken at CRM 10 (^{137}Cs in crappie in 1962-1963; some of the samples in 1981-1985), CRM 12 (1977-1985), CRM 13.3 (1948), CRM 14.5-14.6 (1960), and CRM 4.5-19.1 (1960-1962; 1963). The graphs for CRM 3.5 include samples at CRM 2 (1984-1985), CRM 2.2 (1960), CRM 4 (1978), CRM 4.5-4.6 (1960), and CRM 5 (1978-1991). Measurements from CRM 10-14.6 could have included fish that had recently come from Poplar Creek, and measurements from CRM 2-5 could have included fish that had recently come from the Emory River or the Tennessee River. In addition, because fish such as carp, catfish, and largemouth bass may swim considerable distances, the location at which a fish was caught might not be representative of a location where the fish had recently spent considerable time.

(5) As discussed in Section 8.1.5, the data themselves are not ideal in terms of representativeness (species, sample size, location) or consistency (sampling procedures and preparation in different years, composite vs. individual samples, species and locations sampled in different years). In almost all cases, the samples were taken and measurements made for other reasons than determining the bioconcentration factors of radionuclides in fish. A few of the 1960 samples may represent single fish (Table 8A-2). Any sample of fish flesh that contained a bone fragment would be expected to give a higher measurement for ^{90}Sr than would be found for flesh alone; the likelihood of such an event is dependent on the care taken in sampling preparation, which may not have been consistent from year to year or with various technical personnel. A composite sample might be biased inappropriately if unequal or unweighted portions were taken from individual fish.

In summary, the subjective confidence limits for the predicted annual average concentrations in fish are not expected to contain all available measurements of radionuclide concentrations in fish, nor should the confidence limits be widened to include all the measurements. The fact that, in spite of limitations on the data, about three-fourths of the measurements do fall on or inside the 90% subjective confidence intervals supports the validity of the approach used to predict the annual average radionuclide concentrations in Clinch River fish for the years and locations of interest. Further comparison of predicted and measured concentrations in fish would require estimation of average water concentrations on smaller time scales corresponding to the fish sampling, as well as use of a dynamic model of the uptake and retention of each radionuclide in fish.

8.5 Special Cases

Although this report concentrates on doses and risks from consumption of Clinch River fish, for the sake of completeness, two related issues are discussed here: (1) fish from White Oak Lake or White Oak Creek that might have gone over White Oak Dam and been caught soon after in the Clinch River (e.g., a single "hot" fish); and (2) fish caught in the Tennessee River or Watts Bar Reservoir below the entry of the Clinch River. Many more people might have consumed fish from the Tennessee River or Watts Bar Reservoir than from the Clinch River; additionally, an individual may have consumed a larger quantity of fish from the Tennessee River-Watts Bar Reservoir system than from the Clinch River.

8.5.1 Fish from White Oak Lake

A situation of potential concern is the case of a single fish migrating past White Oak Dam from White Oak Lake and being caught soon thereafter in the Clinch River. While such a situation is not expected to occur frequently, nevertheless the possibility must be considered (see Martin et al., 1964). The present study therefore included dose calculations for a person consuming a single fish from White Oak Lake.

The maximum reported radionuclide concentrations in fish (flesh and bones) in the Clinch River, White Oak Creek, and White Oak Lake are given in Table 8.4. These fish were sampled in 1948 (Clinch River and White Oak Creek) and 1949 (White Oak Lake) (Knobf, 1951; see Section 8.1.1; activity in flesh is primarily ^{137}Cs and in bone, ^{90}Sr). For Upper White Oak Lake, the values are actually the mean concentrations of 3 or 4 fish; the individual fish were not reported. It is possible, of course, that some fish might have exceeded these concentrations, as radionuclide concentrations in water may have been higher at other times than when these fish were sampled. In addition to these fish, 3 fish caught in 1944 were found to have total activities of 1800, 1900, and 220,000 Bq kg^{-1} (49,000, 51,000 and 6,000,000 pCi kg^{-1} in white perch, catfish, and catfish, respectively; Curtis, 1944). These measurements were probably made for the whole fish (including viscera); no istopic composition was reported. Of 19 fish of 16 species caught between CRM 21.7 and CRM 24.4 in 1960 (March 23-April 1), 14 had gross gamma activities greater than 1 count per minute per gram; the highest was a bluegill with 14.7 counts per minute per gram (based on ORNL/TVA, 1960). Using the estimated conversion described in Section 8.1.1 that was used for

Knobf's (1951) data, this would be equivalent to 66,000 pCi kg^{-1} or 2400 Bq kg^{-1}. This activity was probably measured for the whole fish; no isotopic analysis was reported.

Table 8.4 Highest reported radionuclide concentrations in flesh and bones of fish from the Clinch River, White Oak Creek, and White Oak Lake (derived from Knobf, 1951).

Location	Species	Concentration in Flesh[a] (Bq kg^{-1})	Concentration in Bone[b] (Bq kg^{-1})
Upper White Oak Lake[c]	Redhorse	79,000	360,000
	Carp	37,000	470,000
Lower White Oak Lake	Crappie	34,000	260,000
White Oak Creek	Bass	18,000	110,000
	Sunfish	8,100	460,000
CRM 18-19	Crappie	4,500	76,000
CRM 13.3	Crappie	3,300	13,000

[a] Activity in flesh was primarily ^{137}Cs.
[b] Activity in bone was primarily ^{90}Sr.
[c] Values are averages of 3 (carp) or 4 (redhorse) individual fish.

8.5.2 Fish in the Tennessee River and Watts Bar Reservoir

Many more people might have consumed fish from the Tennessee River or Watts Bar Reservoir than from the Clinch River; additionally, an individual may have consumed a larger quantity of fish from the Tennessee River-Watts Bar Reservoir system than from the Clinch River. In general, however, radionuclide concentrations in fish in the Tennessee River and Watts Bar Reservoir are expected to be considerably lower than for the Clinch River, due to the additional dilution afforded by the Tennessee River. Fish swimming into the Tennessee from the Clinch are expected to have the highest concentrations of radionuclides of any fish in the Tennessee, but these fish are few in number in comparison to the fish already in the Tennessee. Table 8.5 gives a compilation of available information on radionuclide concentrations in fish in the Tennessee River system. In most cases, concentrations do not differ much with increasing distance downstream; a consistent downward trend in the concentrations with increasing downstream distance is seen in some samples but not in others.

Comparison of measured concentrations in fish in the Clinch and Tennessee Rivers shows that mean concentrations of ^{137}Cs in Tennessee River fish may be a factor of 4-25 lower than in Clinch River fish for the same time periods. Mean concentrations in Tennessee River fish may be a factor of 1.5-10 lower for ^{90}Sr and 1.5-7 lower for ^{106}Ru. For ^{60}Co, mean concentrations in Tennessee River fish range from a factor

of 3 lower than Clinch River fish to about the same to a factor of 2 higher. Bioconcentration factors for Tennessee River fish were not determined.

Table 8.5 Historical average measured concentrations of radionuclides in fish in the Tennessee River.

Time Period	Location	Species	Radionuclide	Average Concentration (Bq kg^{-1})	References
1960	TRM 562.7	bottom-feeders	^{137}Cs	2.8	A
			^{90}Sr (flesh)	5.6	
			^{90}Sr (whole)	110	
			^{106}Ru	1.1	
			^{60}Co	3.0	
	TRM 517.9	bottom-feeders	^{137}Cs	2.4	
			^{90}Sr (flesh)	5.2	
			^{90}Sr (whole)	190	
			^{106}Ru	0.56	
			^{60}Co	10	
	TRM 417.0	bottom-feeders	^{137}Cs	1.1	
			^{90}Sr (flesh)	6.5	
			^{90}Sr (whole)	32	
			^{106}Ru	1.3	
			^{60}Co	1.5	
1960-62	Tennessee River	carp	^{137}Cs	6.7	B
			^{90}Sr (flesh)	4.4	
			^{106}Ru	3.0	
			^{60}Co	2.6	
		carpsucker	^{137}Cs	4.8	
			^{90}Sr (flesh)	3.7	
			^{106}Ru	2.6	
			^{60}Co	2.3	
		sight-feeders	^{137}Cs	6.3	
			^{90}Sr (flesh)	9.3	
			^{106}Ru	1.8	
			^{60}Co	2.4	
1963	Tennessee River	carp	^{137}Cs	2.3	B
			^{90}Sr (flesh)	0.19	

Table 8.5 (Continued)

Time Period	Location	Species	Radionuclide	Average Concentration ($Bq\ kg^{-1}$)	References
		buffalo	^{137}Cs	2.7	
			^{90}Sr (flesh)	0.33	
1977-91	Watts Bar Reservoir	catfish, crappie, buffalo	^{137}Cs	0-9.3	C
	Chickamauga and Nickajack Reservoirs	catfish, crappie, buffalo	^{137}Cs	0-5.6	C
	not given	bluegill, catfish, bass	^{137}Cs	5.9 (maximum)	D
	not given	bluegill, catfish, bass	^{90}Sr	0.37 (maximum)	D
1989-90	TRM 557.0	not given	^{137}Cs	3.3 (maximum)	E
	TRM 530.5	not given	^{137}Cs	2.6 (maximum)	E
	TRM 518.0	not given	^{137}Cs	3 (maximum)	E

References:

A Morton, 1962
B Cowser and Snyder, 1966
C TVA, 1992 (as cited in USDOE, 1995)
D USDOE, 1995
E ORNL, 1992

8.6 Summary

Bioconcentration factors for ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co were developed to permit the estimation of radionuclide concentrations in Clinch River fish as a function of the radionuclide concentrations in the river water (concentration in water × bioconcentration factor = concentration in fish). The uncertainty in each mean bioconcentration factor was described as a log-triangular distribution consisting of alternative possible mean values. The central, minimum, and maximum values are given by radionuclide in Table 8.2.

For a human receptor located at CRM 20.5 (where the plume is not completely mixed), an adjustment factor was developed to permit estimation of fish concentrations at CRM 20.5 from the predicted concentrations at CRM 14. This adjustment factor was described as a log-triangular distribution with a central value of 6 and minimum and maximum values of 1 and 30, respectively.

For calculations of doses and risks from the consumption of fish patties (ground fish including bones), an adjustment factor for ^{90}Sr was developed to permit estimation of the radionuclide concentration in the fish patties from the predicted concentration in fish flesh. The adjustment factor for ^{90}Sr was described as a log-triangular distribution with a central value of 4 and minimum and maximum values of 1 and 24, respectively. Doses and risks from consumption of fish patties are discussed in Section 12.

For calculations of doses and risks for a reference individual catching and consuming a single, highly contaminated fish migrating from White Oak Lake, a summary of the highest reported concentrations in flesh and bones is provided in Table 8.4. Doses and risks from consumption of a singe "hot" fish are discussed in Section 12.

Calculations were not performed in this study for individuals consuming fish from the Tennessee River or Watts Bar Reservoir. Calculations carried out for Clinch River fish provide a reasonable upper bound on the doses and risks for most individuals consuming fish from the Tennessee River or Watts Bar Reservoir, taking into account differences in consumption patterns. In general, mean radionuclide concentrations in fish appear to run a factor of 4-25 lower for Tennessee River/Watts Bar Reservoir fish than Clinch River fish for ^{137}Cs, a factor of 1.5-10 lower for ^{90}Sr, a factor of 1.5-7 lower for ^{106}Ru, and between a factor of 3 lower and 2 higher for ^{60}Co. Bioconcentration factors for Tennessee River fish were not determined.

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9.0 TRANSFER OF SELECTED RADIONUCLIDES FROM RIVER WATER TO MILK AND BEEF

Section 9 describes the calculations and assumptions used to estimate the transfer of radionuclides to milk and beef consumed by humans. Radionuclides released into the Clinch River from Oak Ridge National Laboratory could be ingested by grazing dairy and beef cattle if these cattle used the river as a source of drinking water. Once ingested, these radionuclides are absorbed by the GI tract and transferred through the bloodstream to milk and animal muscle. Estimation of the specific factors (with corresponding uncertainty) that describe the transfer of radionuclides to milk and to beef are discussed in detail in this section. In addition, a sensitivity analysis has been performed to indicate which input parameters contribute the most uncertainty in the overall transfer of radionuclides from river water to milk and beef.

For the calculations in this section, all animals are assumed to be dairy and beef cattle raised by their owners primarily for family use. Other types of farm animals such as goats and pigs are likely to have had access to sources of drinking water other than the river, so that contamination by infrequent consumption of river water is considered negligible (see Section 7.2.1). During the period of time when the largest radionuclide releases occurred (1944 to the 1960s), the primary breed of dairy cattle in East Tennessee used for milk by farm families was the Jersey cow, which produced approximately 1 to 2 gallons per day (Miller, 1996). Hereford cattle were used as the main source of beef for farm families (Prichard, personal communication with C. Lewis, 1997). Section 9 specifically addresses (1) the estimation of the potential transfer of ^{137}Cs, ^{90}Sr, ^{60}Co, and ^{131}I from river water to milk (milk ingestion exposure pathway) and (2) the estimation of the potential transfer of ^{137}Cs, ^{106}Ru, ^{90}Sr, and ^{60}Co from river water to meat (meat ingestion exposure pathway).

9.1 Modeling Approach

The following equations are used to estimate the transfer of selected radionuclides from ingested water to milk or to beef.

$$WM = \frac{C_m}{C_w} = Q_m \cdot P_m \cdot F_m \qquad \text{(Eq. 9.1)}$$

where:

WM = the water-to-milk transfer factor [Bq L^{-1}_{milk} per Bq L^{-1}_{water}];

C_m = the concentration of radionuclide in milk [Bq L^{-1}];

C_w = the concentration of radionuclide in water [Bq L^{-1}];

Q_m = the water ingestion rate for dairy cattle [L d^{-1}];

P_m = the fraction of drinking water that is contaminated for dairy cattle [unitless], and

F_m = the intake-to-milk transfer coefficient for dairy cattle [d L^{-1}].

$$WF = \frac{C_f}{C_w} = Q_f \cdot P_f \cdot F_f \qquad \text{(Eq. 9.2)}$$

where:

WF = the water-to-beef transfer factor [Bq kg^{-1} per Bq L^{-1}];

C_f = the concentration of radionuclide in beef [Bq kg^{-1}];

C_w = the concentration of radionuclide in water [Bq L^{-1}];

Q_f = the water ingestion rate for beef cattle [L d^{-1}];

P_f = the fraction of drinking water that is contaminated for beef cattle [unitless], and

F_f = the intake-to-meat transfer coefficient for beef cattle [d kg^{-1}].

9.2 Input Parameters

The ranges and distributions used to describe the state of knowledge about true but unknown values for the parameters used for derivation of the intake of contamination from river water by milk and beef cattle (Table 9.1) were specified based on judgment after review of the literature. The rationales for the choice of the specific parameter values for the average amount of water ingested on a daily basis (Q_m and Q_f) and for the fraction of drinking water for cattle that is contaminated (P_m and P_f) are discussed in Section 7.8.3.

Table 9.1 Parameter values used to calculate the intake of radionuclides from water by milk and beef cattle.

Parameter	Units	Distribution Minimum	Distribution Maximum	Shape	Rationale
Dairy cattle					
Q_m	L d^{-1}	32	60	Uniform	See Section 7.8.3
P_m	unitless	0.05	0.25	Uniform	See Section 7.8.3
Beef cattle					
Q_f	L d^{-1}	20	50	Uniform	See Section 7.8.3
P_f	unitless	0.5	1.0	Uniform	See Section 7.8.3

9.2.1 Transfer of Radionuclides from a Daily Intake to Milk and Beef

The transfer of ingested radionuclides to animal products is commonly described using the transfer coefficients F_m for milk and F_f for meat (flesh). In the following sections, the rationales are given for the selection of ranges and distributions describing the milk and beef transfer coefficients.

9.2.1.1 Milk Transfer Coefficient

The milk transfer coefficient (F_m) represents the fraction of the total daily intake of a radionuclide that is transferred to one liter of a cow's milk at equilibrium. The radionuclides of primary interest for the milk ingestion pathway are ^{137}Cs, ^{90}Sr, ^{60}Co, and ^{131}I.

Cesium-137

Numerous studies report an expected or "most likely" value for use in modeling the transfer of ^{137}Cs to cow's milk. The most commonly reported F_m for ^{137}Cs for use in different radiological assessment codes and reports is on the order of 7.0×10^{-3} d L^{-1} (Ng, 1982; Baes et al., 1984; Summerling et al., 1984; Napier et al., 1988; Kennedy and Strenge, 1992; Peterson, 1983). The International Atomic Energy Agency's handbook of parameter values for the prediction of radionuclide transfer (IAEA, 1994) lists an expected value of 7.9×10^{-3} d L^{-1}, with a minimum value of 1.0×10^{-3} and a maximum of 2.7×10^{-2}. In a document that addressed the variability in dose estimates associated with food chain transport (Hoffman et al., 1982), a lognormal distribution was specified with a geometric mean of 6.7×10^{-3} d L^{-1} and geometric standard deviation of 1.79 (95% confidence interval from 2.1×10^{-3} to 2.1×10^{-2}). The lognormal distribution and range given in Hoffman et al. (1982) were used in a later report that estimated and ranked potential human risks associated with contaminants discharged from the waste area groupings at Oak Ridge National Laboratory (Shevenell and Hoffman, 1993).

In a joint effort by the U.S. Nuclear Regulatory Commission and the Commission of European Communities, sixteen experts from eight countries were selected to give their professional evaluation (expert

elicitation) of the uncertainty associated with the estimation of the transfer of radionuclides from soil to plants and plants to animal food products (Brown et al., 1997). Ten experts participated in the estimate of transfer factors for radionuclides to animal food products (Table 9.2). Table 9.3 compares the 5%, 50%, and 95% subjective confidence levels recommended by each individual expert for use in uncertainty analysis of a regional average resulting from an accidental release from a nuclear power plant. The reported aggregate expert distribution, resulting from equal weighting of continuous distribution functions fitted to the experts' stated objective probability estimates, ranged from 1.0×10^{-3} to 2.4×10^{-2} d L^{-1} (90% confidence interval), with 5.7×10^{-3} given as the median value for the transfer of ^{137}Cs to milk.

When analyzing and compiling the responses from the experts, Brown et al. (1997) chose to fit a minimum informative distribution that satisfies the quantile information of the experts. In using the concept of minimum information, a so-called background measure, to which the information of the resulting distribution must be minimal, has to be specified. Brown et al. (1997) used the uniform and log-uniform distribution as the background measure. However, this methodology neglects central value knowledge given by each expert. Therefore, the raw data submitted by each expert were analyzed further in this assessment. Each expert's recommendation was weighted equally, and a log-triangular distribution was chosen to describe the quantile information given by each expert. In addition, this assessment has included the responses of two additional experts which Brown et al. (1997) chose to neglect. Their reasoning for not including the two experts' recommendations is that their responses were not complete. However, these two experts did give responses for the transfer of ^{137}Cs to milk. Therefore, they were included in this assessment. Inclusion of all ten experts yields a 90% subjective confidence interval that ranges from 9.2×10^{-4} to 2.7×10^{-2} d L^{-1} with a median of 5.4×10^{3}. The resulting values are best characterized using a lognormal distribution with a geometric mean of 5.4×10^{-3} d L^{-1} and a geometric standard deviation of 2.7.

Table 9.2 Food chain experts elicited by the U.S. Nuclear Regulatory Commission and the Commission of European Communities (Brown et al., 1997).

Expert	Country
Peter Coughtrey	UK
Francois Daburom	France
F. Owen Hoffman	US
F. Ward Wicker	US
Brenda Howard	UK
Jack Pearce	UK
Per Strand	Norway
Christian Vandecasteele	Belgium
Gabriel Voigt	Germany
Gerald Ward	US

Table 9.3 Subjective probability estimates of uncertainty in milk transfer coefficients for ^{137}Cs recommended by individual experts (Brown et al., 1997).

Unidentified NRC/CEC Expert	F_m (d L^{-1}) 5%	50%	95%
H	2.0×10^{-3}	7.0×10^{-3}	2.0×10^{-2}
I	1.0×10^{-3}	3.0×10^{-3}	1.5×10^{-2}
J	1.0×10^{-3}	8.0×10^{-3}	3.0×10^{-2}
K	2.0×10^{-3}	8.0×10^{-3}	2.0×10^{-2}
L	1.2×10^{-3}	4.4×10^{-3}	7.6×10^{-3}
M[a]	5.0×10^{-3} [b]	2.0×10^{-3}	5.0×10^{-3}
N	1.5×10^{-4}	7.0×10^{-3}	3.0×10^{-2}
O[a]	3.0×10^{-3}	6.0×10^{-3}	1.0×10^{-2}
P	1.7×10^{-3}	5.0×10^{-3}	2.0×10^{-2}
Q	2.0×10^{-3}	8.0×10^{-3}	2.0×10^{-1} [c]
Aggregated results			
NRC/CEC	1.0×10^{-3}	5.7×10^{-3}	2.4×10^{-2}
This assessment	9.2×10^{-4}	5.4×10^{-3}	2.7×10^{-2}

[a] This expert's results were not included in the NRC/CEC aggregated results.
[b] Misprint in the report (5.0×10^{-4} is the intended value).
[c] Misprint in the report (2.0×10^{-2} is the intended value).

Strontium-90

The most commonly assumed F_m value found in the literature for the assessment of ^{90}Sr is on the order of 1.5×10^{-3} d L^{-1} (Baes et al., 1984; Napier et al., 1988; Kennedy and Strenge, 1992; Peterson, 1983). The International Atomic Energy Agency's handbook of parameter values for the prediction of radionuclide transfer (IAEA, 1994) lists an expected value of 2.8×10^{-3} d L^{-1}, with a minimum value of 1.0×10^{-3} and a maximum of 3.0×10^{-3}. Hoffman et al. (1982) suggest the use of a lognormal distribution with a geometric mean of 1.2×10^{-3} d L^{-1} and a geometric standard deviation of 1.62 (95% confidence interval ranging from 4.6×10^{-4} to 3.0×10^{-3}).

The subjective 5^{th}, 50^{th}, and 95^{th} percentiles recommended by each of the ten individual experts that participated in the NRC/CEC formal elicitation are given in Table 9.4. The reported aggregate result from eight experts for the transfer of ^{90}Sr to milk is a 90% subjective confidence interval that ranges from 4.3×10^{-4} to 4.8×10^{-3} d L^{-1}, with 2.3×10^{-3} d L^{-1} given as the median value. This aggregated result is obtained by first fitting a cumulative distribution function to the percentiles given by each expert and then sampling from all eight distributions with equal weight. This method provides additional information not produced by the individual experts. For this reason, equal weighting is assigned to the values specified at each given subjective probability level, expressed as an overall mean value for the 5^{th}, 50^{th}, and 95^{th} percentiles (Table 9.4).

As discussed for ^{137}Cs, two additional experts' responses were included in the aggregated result obtained in this assessment, based on equal weighting and a log-triangular distribution for each of the expert's recommendations. The 90% subjective confidence interval ranges from 4.5×10^{-4} to 4.4×10^{-3} d L^{-1} with a median of 2.3×10^{-3}. These results are best fitted to a lognormal distribution with a geometric mean of 1.9×10^{-3} d L^{-1} and a geometric standard deviation of 2.0.

Table 9.4 Subjective probability estimates of uncertainty in milk transfer coefficients for ^{90}Sr recommended by individual experts (Brown et al., 1997).

Unidentified NRC/CEC Expert	F_m (d L^{-1}) 5%	50%	95%
H	1.0×10^{-3}	3.0×10^{-3}	5.0×10^{-3}
I	4.0×10^{-4}	2.0×10^{-3}	5.0×10^{-3}
J	1.0×10^{-3}	2.0×10^{-3}	3.0×10^{-3}
K	5.0×10^{-4}	2.8×10^{-3}	4.0×10^{-3}
L	9.0×10^{-4}	2.1×10^{-3}	4.3×10^{-3}
M[a]	5.0×10^{-4}	1.0×10^{-3}	7.0×10^{-3}
N	3.0×10^{-4}	1.2×10^{-3}	5.0×10^{-3}
O[a]	5.0×10^{-4}	1.0×10^{-3}	3.0×10^{-3}
Q	1.0×10^{-3}	3.0×10^{-3}	4.0×10^{-3}
P	3.0×10^{-4}	1.7×10^{-3}	5.0×10^{-3}
Aggregated results			
NRC/CEC	4.3×10^{-4}	2.3×10^{-3}	4.8×10^{-3}
This assessment	4.5×10^{-4}	2.3×10^{-3}	4.4×10^{-3}

[a] This expert's results were not included in the NRC/CEC aggregated results.

Cobalt-60

The most commonly assumed value in radiological assessment models for F_m for ^{60}Co is on the order of 2.0×10^{-3} d L^{-1} (Baes et al., 1984; IAEA, 1982; Kennedy and Strenge, 1992; Peterson, 1983). The NRC/CEC panel of experts did not include ^{60}Co among the radionuclides of concern. Hoffman et al. (1984), in a document comparing predictions from internationally recognized assessment models for the transfer of radionuclides through the terrestrial food chain, reports a 95% subjective confidence interval of 1.2×10^{-4} d L^{-1} to 1.2×10^{-2} d L^{-1}, with a geometric mean of 1.2×10^{-3}. The International Atomic Energy Agency's handbook of parameter values for the prediction of radionuclide transfer (IAEA, 1994) reports a minimum value of 6.0×10^{-5} d L^{-1} and a maximum of 1.0×10^{-2} d L^{-1}. A review of background information indicates that these values have been selected to err on the side of conservatism and are biased high due to the potential for high accumulation of organic forms of cobalt. The IAEA states that such a wide range exists because cobalt transfer differs according to its chemical form. For organically bound cobalt, a value of 3.0×10^{-4} is expected, whereas for inorganic forms, a value of 7.0×10^{-5} is more appropriate.

Because ^{60}Co released from White Oak Lake was most likely to be in the inorganic form, the minimum F_m value reported in IAEA's Handbook of Parameter Values (6.0×10^{-5} d L^{-1}) is selected in this report as the minimum value of a log-uniform distribution. A value of 1.0×10^{-2} d L^{-1} was chosen as a maximum upper bound to encompass the current state of knowledge for the transfer of various compounds of ^{60}Co from the diet to the milk of dairy cows.

Iodine-131

Extensive literature documents the variability of measured values for the F_m for radioiodine. In one literature review, Hoffman (1978) reported a range of 4×10^{-3} to 1.0×10^{-2} d L^{-1}. In a statistical analysis of average values among nineteen publications, Hoffman and Baes (1979) reported a lognormal distribution with a geometric mean of 1×10^{-2} d L^{-1} and a geometric standard deviation of 1.73, which produces a 95% confidence interval ranging from 3.4×10^{-3} to 2.9×10^{-2} d L^{-1}. As a result of environmental monitoring in the aftermath of the Chernobyl accident, Köhler et al. (1991) reported values of F_m from eight locations in the Northern Hemisphere ranging from 1×10^{-3} to 7.3×10^{-3} d L^{-1}.

For the Hanford Environmental Dose Reconstruction, which involved the investigation of releases of ^{131}I from 1944 to 1956 at the Hanford site in eastern Washington, a lognormal distribution was assigned to the F_m for individual backyard cows with a geometric mean of 9.2×10^{-3} d L^{-1}, a geometric standard deviation of 2.1, and a range from 1.6×10^{-3} to 5.2×10^{-2} d L^{-1}. For a herd of commercial cows, a normal distribution was assigned with a mean of 1.2×10^{-2} d L^{-1}, a standard deviation of 0.002 d L^{-1}, and a range extending from 7.3×10^{-3} to 1.6×10^{-2} (Snyder et al., 1994).

In review of 81 reported values, the National Cancer Institute (1997) described the variability in F_m for ^{131}I as a lognormal distribution with a geometric mean of 4.4×10^{-3} d L^{-1}, a geometric standard deviation of 2.1, and a range of 4×10^{-4} to 2.1×10^{-2} d L^{-1}.

The results of the NRC/CEC expert elicitation are similar to those presented in previous literature reviews. Based on equal weighting of cumulative probability distributions that were fit to the values given by eight experts at the 5^{th}, 50^{th}, and 95^{th} percentiles, the aggregate of the 90% subjective confidence range was 5.3×10^{-4} to 3.7×10^{-2} d L^{-1} (Table 9.5). In this assessment, using equal weighting and log-triangular distributions for all ten experts' results, a 90% subjective confidence interval of 1.6×10^{-3} to 2.7×10^{-2} was obtained, with a median value of 7.0×10^{-3} (Table 9.5).

Table 9.5 Subjective probability estimates of uncertainty in milk transfer coefficients for ^{131}I recommended by individual experts (Brown et al., 1997).

Unidentified NRC/CEC Expert	F_m (d L^{-1}) 5%	50%	95%
H	1.0×10^{-3}	1.0×10^{-2}	4.0×10^{-2}
I	1.0×10^{-3}	7.0×10^{-3}	3.0×10^{-2}
J	1.0×10^{-3}	1.0×10^{-2}	4.0×10^{-2}
K	1.0×10^{-3}	1.0×10^{-2}	3.5×10^{-2}
L	1.0×10^{-3}	1.1×10^{-2}	3.4×10^{-2}
M[a]	1.0×10^{-3}	5.0×10^{-3}	1.0×10^{-2}
N	5.0×10^{-4}	5.0×10^{-3}	5.0×10^{-2}
O[a]	2.0×10^{-3}	4.0×10^{-3}	1.0×10^{-2}
P	2.0×10^{-3}	4.0×10^{-3}	1.8×10^{-2}
Q	2.0×10^{-3}	1.0×10^{-2}	2.0×10^{-2}
Aggregated results			
NRC/CEC	5.3×10^{-4}	7.6×10^{-3}	3.7×10^{-2}
This assessment	1.6×10^{-3}	7.0×10^{-3}	2.7×10^{-2}

[a] This expert's results were not included in the NRC/CEC aggregated results.

In local investigations on the transfer of ^{131}I into the milk of dairy cows, Dr. J.K. Miller (1996) at the University of Tennessee, Knoxville, found an inverse relationship between milk yield and iodine concentration in milk. This finding is important for the present study because backyard cows are considered to be low producers in terms of a daily yield of milk. From unpublished data provided by Miller (1996), values of F_m for cows whose milk production was less than 10 L d^{-1} were investigated. The distribution of these values is lognormal with a geometric mean of 9×10^{-3} d L^{-1} and a geometric standard deviation of 1.9. The values for this distribution come from 77 measurements of lactating Jersey and Holstein cows. This distribution is applied both in this Task 4 report and in Task 1 (releases of ^{131}I to the atmosphere from the processing of radioactive lanthanum at the X-10 facility) to describe the current state of knowledge of the F_m for the transfer of ^{131}I into the milk of backyard cows. This distribution[1] is similar to those described in previous reports, including the results of the recent NRC/CEC expert elicitation (Brown et al., 1997).

[1] In Task 1, a different distribution will be used for describing the uncertainty in the milk transfer coefficient for ^{131}I into the milk of commercial dairy cows because these animals typically produce more than 10 L of milk per day and the milk is pooled from the production of a large population of animals. Thus the effect of uncertainty due to inter-cow variability of F_m is reduced considerably (see Hoffman et al., 1997).

Summary

A summary of the distributions used for the intake-to-milk transfer coefficient is found in Table 9.6.

Table 9.6 A summary of distributions used for the intake-to-milk transfer coefficient (F_m, d L^{-1}) for dairy cattle.

Radionuclide	Parameter 1	Parameter 2	Shape	Primary Reference
^{137}Cs[a]	5.4×10^{-3}	2.7	log-triangular	Brown et al., 1997
^{90}Sr[a]	1.9×10^{-3}	2.0	log-triangular	Brown et al., 1997
^{60}Co[b]	6.0×10^{-5}	1.0×10^{-2}	log-uniform	IAEA, 1994
^{131}I[a]	9.0×10^{-3}	1.9	log-normal	Hoffman et al., 1997; Miller, 1996

[a] Parameters 1 and 2 represent the geometric mean and standard deviation, respectively.
[b] Parameters 1 and 2 represent the minimum and maximum of a log-uniform distribution.

9.2.1.2 Meat Transfer Coefficient

The meat transfer coefficient (F_f) represents the fraction of the total daily intake of a radionuclide that is transferred to one kilogram of animal flesh at equilibrium or at the time of slaughter. The radionuclides of primary interest for the beef ingestion pathway are ^{137}Cs, ^{106}Ru, ^{90}Sr, and ^{60}Co.

The transfer coefficients for meat are more difficult to obtain experimentally than the transfer coefficients for milk because fewer data are available (Ng, 1982; Ng et al., 1982). Furthermore, when data exist, they are most likely for animals other than for beef cattle because the animal must be slaughtered to obtain the estimate of the transfer coefficient. Chickens, a much smaller meat producer, are often used as a surrogate for beef cattle because a larger number can be easily used during the experiment. To use data not specific for cattle, various assumptions must be made about the similarity in the meat-to-feed concentration ratio for different species, about the feed consumption rate and total body mass and mass of muscle tissue in different species, and about the similarity in the uptake and retention pattern of chemically related elements (Ng, 1982). Although F_f values for ^{137}Cs and ^{90}Sr are relatively well documented for some animals, data are sparse for ^{60}Co and ^{106}Ru. Therefore, considerable uncertainty exists in the transfer factors for these two radionuclides. The rationale for the ranges and distributions selected for F_f for the four radionuclides (^{137}Cs, ^{106}Ru, ^{90}Sr and ^{60}Co) follows.

Cesium-137

Several studies report an expected or "most likely" value for the transfer of ^{137}Cs to beef. The most commonly assumed default value of F_f in radiological assessment models for ^{137}Cs is on the order of 2.0×10^{-2} d kg^{-1} (Baes et al., 1984; IAEA, 1994; Kennedy and Strenge, 1992; Ng et al., 1982).

The International Atomic Energy Agency's handbook of parameter values for the prediction of radionuclide transfer (IAEA, 1994) lists an expected value of 5.0×10^{-2} d kg^{-1}, with a minimum value of 1.0×10^{-2} and a maximum of 6.0×10^{-2}. In a document dealing with variability in dose estimates associated with food chain transport (Hoffman et al., 1982), a log-normal distribution with geometric mean 2.1×10^{-2} d kg^{-1} and geometric standard deviation 2.0 is recommended. Also, a minimum (3.0×10^{-3} d kg^{-1}) and maximum (2.0×10^{-1} d kg^{-1}) are given as levels of truncation. Hoffman et al. (1984) give a 95% confidence interval ranging from 2.1×10^{-3} to 2.1×10^{-2} d kg^{-1}.

The recent NRC/CEC expert elicitation (Brown et al., 1997) produced an aggregated distribution for F_f for ^{137}Cs ranging from 3.1×10^{-3} to 9.1×10^{-2} d kg^{-1} (90% subjective confidence interval) with a median value of 4.0×10^{-2}, based on equal weighting of eight expert contributions (Table 9.7).

The method of equal weighting employed in this assessment (discussed in Section 9.2.1.1; Cesium-137) yields the following values for the 5th, 50th, and 95th percentiles, respectively: 1.2×10^{-2}, 4.4×10^{-2}, and 7.6×10^{-2} d L^{-1} (Table 9.7). Data-fitting techniques suggest that these data are best characterized by a lognormal distribution (geometric mean = 3.7×10^{-2} d L^{-1}; geometric standard deviation = 1.8). This distribution best describes the current state of knowledge with respect to uncertainty in the true value for F_f for ^{137}Cs.

Table 9.7 Subjective probability estimates of uncertainty in meat transfer coefficients for ^{137}Cs recommended by individual experts (Brown et al., 1997).

Unidentified NRC/CEC Expert	F_f (d kg^{-1}) 5%	50%	95%
H	8.0×10^{-3}	5.0×10^{-2}	7.0×10^{-2}
I	1.0×10^{-2}	4.0×10^{-2}	1.0×10^{-1}
J	1.0×10^{-2}	5.0×10^{-2}	7.0×10^{-2}
K	1.2×10^{-2}	5.0×10^{-2}	5.8×10^{-2}
L	2.8×10^{-3}	3.0×10^{-2}	5.7×10^{-2}
M[a]	1.0×10^{-3}	3.0×10^{-2}	8.0×10^{-2}
N	8.0×10^{-3}	3.0×10^{-2}	1.2×10^{-1}
O[a]	5.0×10^{-3}	2.0×10^{-2}	6.0×10^{-2}
P	7.0×10^{-3}	3.7×10^{-2}	8.5×10^{-2}
Q	2.0×10^{-2}	5.0×10^{-2}	6.0×10^{-2}
Aggregated results			
NRC/CEC	3.1×10^{-3}	4.0×10^{-2}	9.1×10^{-2}
This assessment	1.2×10^{-2}	4.4×10^{-2}	7.6×10^{-2}

[a] This expert's results were not included in the NRC/CEC aggregated results.

Strontium-90

Based upon the large amount of data found in the literature for the F_f for ^{90}Sr, the knowledge base would appear to warrant a fairly tight range. However, most reported ranges span more than two orders of magnitude. Most transfer factors found in the literature are a result of the derivation of default values for application in regulatory model calculations.

The most commonly reported default value of F_f for ^{90}Sr found in the literature is on the order of 8.0×10^{-4} d kg^{-1} (Napier et al., 1988; Ng et al., 1982; Peterson, 1983). The International Atomic Energy Agency's handbook of parameter values for the prediction of radionuclide transfer (IAEA, 1994) lists an expected value of 8.0×10^{-3}, with a minimum value of 3.0×10^{-4} and a maximum of 8.0×10^{-3}. Hoffman et al. (1983) assumed a geometric mean and geometric standard deviation of 5.8×10^{-4} d kg^{-1} and 3.3, respectively (corresponding to a 95% confidence interval of 7.8×10^{-5} to 1.8×10^{-3} d kg^{-1}).

Information from the formal expert elicitation sponsored by the U.S. Nuclear Regulatory Commission and the Commission of European Communities (Brown et al., 1997) produced a 90% subjective confidence interval for the transfer of ^{90}Sr to beef from 1.6×10^{-4} to 6.2×10^{-2} d kg^{-1}, with a median value of 4.8×10^{-3} d kg^{-1}, based on equal weighting of fitted cumulative distribution functions to the values submitted by each of eight experts. The upper bound value is consistent with the input data given by all but one of the experts (Expert I; Table 9.8).

The method of equal weighting used in this assessment (discussed in Section 9.2.1.1; Cesium-137), yields a 90% subjective confidence interval ranging from 1.5×10^{-4} to 3.2×10^{-2} d L^{-1}, with a median of 3.8×10^{-3}. These data are best described using a lognormal distribution with a geometric mean of 2.6×10^{-3} d L^{-1} and a geometric standard deviation of 5.0. This unusually high geometric standard deviation is a result of expert J's submitted uncertainty range. His rationale for the very large upper bound value is that it includes young cattle, which should have higher transfer coefficients than older cattle.

Table 9.8 Subjective probability estimates of uncertainty in meat transfer coefficients for ^{90}Sr recommended by individual experts (Brown et al., 1997).

Unidentified NRC/CEC Expert	F_f (d kg^{-1})		
	5%	50%	95%
H	1.0×10^{-3}	8.0×10^{-3}	9.0×10^{-3}
I	1.0×10^{-4}	3.0×10^{-4}	8.0×10^{-4}
J	3.0×10^{-4}	8.0×10^{-3}	1.0×10^{-1}
K	2.0×10^{-4}	8.0×10^{-3}	1.4×10^{-2}
L	3.0×10^{-4}	4.2×10^{-3}	8.0×10^{-3}
M[a]	NR[b]	3.0×10^{-4}	NR
N	1.0×10^{-4}	8.0×10^{-4}	6.0×10^{-3}
O[a]	1.0×10^{-4}	6.0×10^{-4}	4.0×10^{-3}
P	3.0×10^{-4}	3.0×10^{-3}	8.0×10^{-3}
Q	5.0×10^{-3}	8.0×10^{-3}	1.0×10^{-2}
Aggregated results			
NRC/CEC	1.6×10^{-4}	4.8×10^{-3}	6.2×10^{-2}
This assessment	1.5×10^{-4}	3.8×10^{-3}	3.2×10^{-2}

[a] This expert's results were not included in the NRC/CEC aggregated results.
[b] NR = no value reported

Cobalt-60

The meat transfer factor for ^{60}Co has much more uncertainty associated with it than do the transfer factors for the other radionuclides in this report, partly because the biological distribution of cobalt is not very well known.

The most common default F_f for ^{60}Co cited in the literature is 2.0×10^{-2} d kg^{-1} (Baes et al., 1984; Napier et al., 1988; Kennedy and Strenge, 1992). Ng et al. (1982) suggest a mean F_f value of 1.3×10^{-2} d kg^{-1}. The same report gives a range of 1.0×10^{-3} to 1.7×10^{-2} from a compilation of literature. Hoffman et al. (1984) report a geometric mean of 9.7×10^{-3} d kg^{-1} and a 95% confidence interval spanning from 1.5×10^{-3} to 6.3×10^{-2}.

The International Atomic Energy Agency's handbook of parameter values for the prediction of radionuclide transfer (IAEA, 1994) reports a minimum value of 4.0×10^{-5} d kg^{-1} and a maximum of 7.0×10^{-2} d kg^{-1}. However, it is noted that this range is wide to account for the transfer of both organic and inorganic forms of cobalt. For organically bound cobalt, an F_f of 1×10^{-2} d kg^{-1} is expected. A lower transfer factor of 1.0×10^{-4} is expected for inorganic forms of cobalt.

Because ^{60}Co found in the river system is assumed to be in an inorganic form, the information reported in the IAEA Handbook of Parameter Values (IAEA, 1994) for inorganic forms of cobalt will be the basis for the development of a range and distribution for this report. The minimum F_f value reported in IAEA's Handbook of Parameter Values (4.0×10^{-5} d kg^{-1}) will be used as the minimum value of a log-uniform distribution. Since IAEA's upper bound value (7.0×10^{-2} d kg^{-1}) is intended to represent the transfer of organic cobalt, a somewhat lower maximum value of 1.0×10^{-2} d kg^{-1} was chosen subjectively to encompass the current state of knowledge for inorganic compounds of ^{60}Co in this report.

Ruthenium-106

Due to an overall lack of information regarding the transfer of ^{106}Ru to meat, the uncertainty associated with the transfer coefficient is large. The most commonly reported default value for the beef transfer coefficient used in mathematical models is on the order of 2.0×10^{-3} d kg^{-1} (NCRP, 1991; Baes et al., 1984; Napier et al., 1988; IAEA, 1982; Kennedy and Strenge, 1992; Ng et al., 1982; Peterson, 1983).

The International Atomic Energy Agency's "Handbook of Parameter Values for the Prediction of Radionuclide Transfer in Temperate Environments" (IAEA, 1994) reports a range of 1.0×10^{-4} to 5.0×10^{-2} d kg^{1}. The same report suggests that the expected value is the same as the upper bound of the range (5.0×10^{-2} d kg^{-1}). In addition, higher ^{106}Ru concentrations are often found in tissues other than muscle, particularly liver.

Due to the lack of concentrated studies on the transfer of ^{106}Ru from intake to beef, we feel that the range provided in IAEA's Handbook of Parameter Values appears to best represent the current state of knowledge about this unknown quantity. The distribution we have selected for use in this project is log-triangular, with most values occurring near the upper end of the distribution. The F_f for ^{106}Ru will then range from 1.0×10^{-4} to 5.0×10^{-2} d L^{-1} with the mode being equal to the upper bound.

Summary

A summary of the parameter distributions used for the intake-to-meat transfer coefficient is found in Table 9.9.

Table 9.9 Parameter distributions used for the intake-to-meat transfer coefficient (F_f, d kg^{-1}) for beef cattle.

Radionuclide	Minimum	Median or (Mode)	GSD	Max	Shape	Primary Reference
^{137}Cs		3.7×10^{-2}	1.8		lognormal	Brown et al., 1997
^{90}Sr		2.6×10^{-3}	5.0		lognormal	Brown et al., 1997
^{60}Co	4.0×10^{-5}			1.0×10^{-2}	log-uniform	IAEA, 1994
^{106}Ru	1.0×10^{-4}	(5.0×10^{-2})		5.0×10^{-2}	log-triangular	IAEA, 1994

9.3 Results

The estimates of the 95% confidence interval for the milk transfer factor (WM) and beef transfer factor (WF) obtained from the uncertainty analysis are shown in Tables 9.10 and 9.11 and Figs. 9.1 and 9.2. For both the milk and meat transfer factors, the largest uncertainty resides in the transfer of ^{60}Co to the milk or meat.

Table 9.10. Estimates of the milk transfer factor [Bq L^{-1} (milk) per Bq L^{-1} (water)].

	Milk Transfer Factor, WM 95% Subjective Confidence Interval		
Radionuclide	lower bound	central value[a]	upper bound
Cesium-137	4.3×10^{-3}	3.3×10^{-2}	3.2×10^{-1}
Strontium-90	2.3×10^{-3}	1.2×10^{-2}	5.1×10^{-2}
Cobalt-60	3.1×10^{-4}	5.0×10^{-3}	7.5×10^{-2}
Iodine-131	1.0×10^{-2}	5.6×10^{-2}	2.5×10^{-1}

[a] median

Table 9.11 Estimates of meat transfer factor [Bq kg^{-1} (meat) per Bq L^{-1} (water)].

Radionuclide	**Meat Transfer Factor, WF 95% Subjective Confidence Interval**		
	lower bound	**central value[a]**	**upper bound**
Cesium-137	2.1×10^{-1}	8.4×10^{-1}	3.5×10^{0}
Strontium-90	2.2×10^{-3}	6.0×10^{-2}	1.4×10^{0}
Cobalt-60	8.0×10^{-4}	1.6×10^{-2}	2.6×10^{-1}
Ruthenium-106	5.8×10^{-3}	1.7×10^{-1}	1.5×10^{0}

[a] median

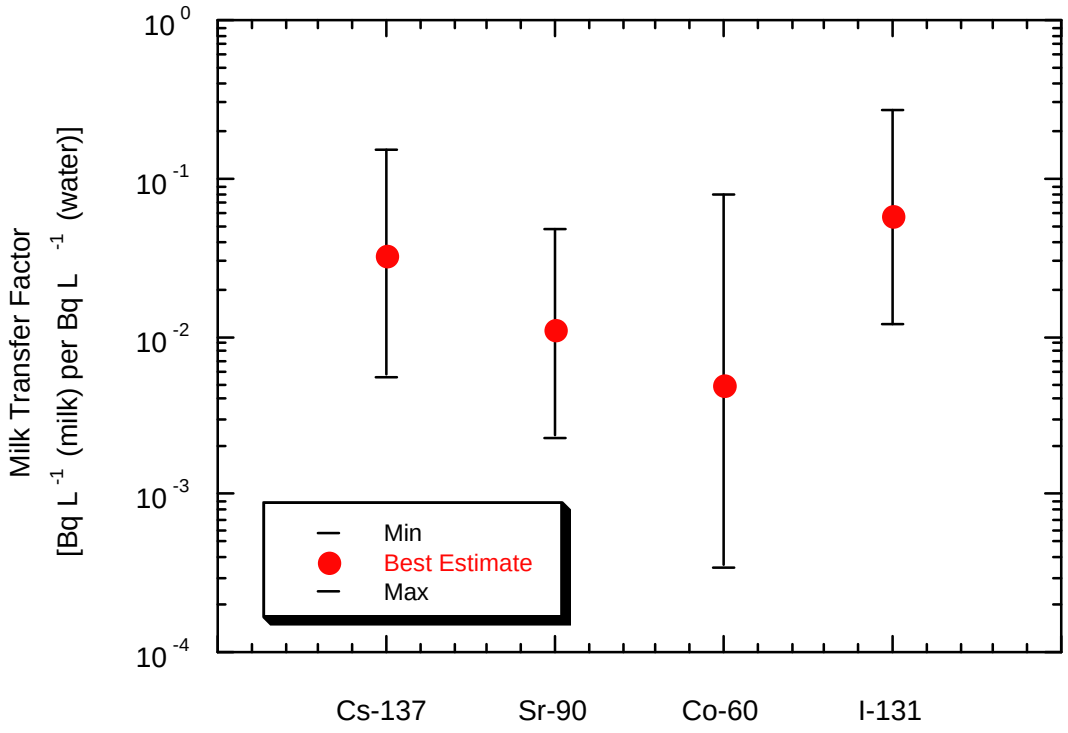

Fig. 9.1 Milk transfer factors (WM) for the radionuclides of concern plotted with their associated 95% subjective confidence intervals.

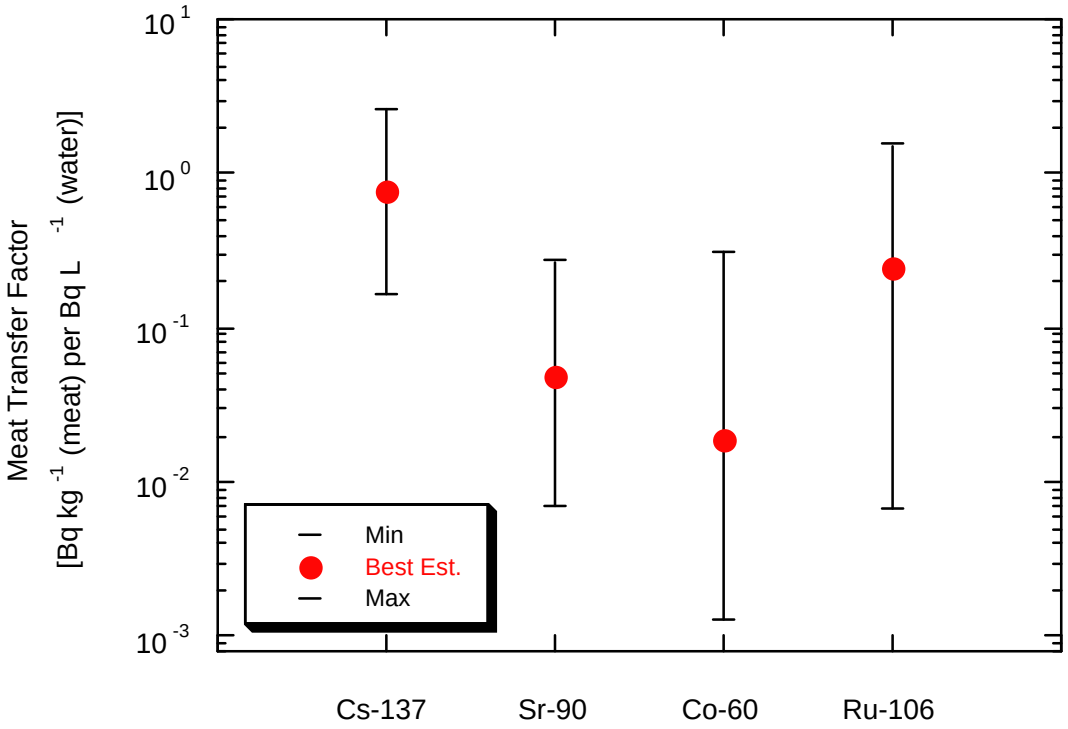

Fig. 9.2 Meat transfer factors (WF) for the radionuclides of concern plotted with their associated 95% subjective confidence intervals.

9.4 Sensitivity Analysis

The uncertainty in the transfer of all radionuclides of concern to milk is dominated by the uncertainty in the intake-to-milk (F_m) transfer coefficient (e.g., ^{131}I results in Table 9.12), while the uncertainties in the water ingestion rates and in the fraction of contaminated water are of minor significance.

The uncertainty in the transfer of ^{137}Cs (and the remaining radionuclides of concern) from river water to meat is also dominated by the intake-to-meat (F_f) transfer coefficient, while the uncertainties in the water ingestion rates and in the fraction of contaminated water are minor contributors (Table 9.13).

Table 9.12 Results of the sensitivity analysis for the milk transfer factor for ^{131}I.

Contribution to the uncertainty in the result (%)	
Parameter	**Water to milk transfer (^{131}I)**
F_m	64.3 %
P_m	29.9 %
Q_m	5.8 %

Table 9.13 Results of the sensitivity analysis for the pasture-meat transfer factor for ^{137}Cs.

Contribution to the uncertainty in the result (%)	
Parameter	**Water to meat transfer (^{137}Cs)**
F_f	64.3 %
P_f	25.8 %
Q_f	9.9 %

9.5 Summary

Subjective probability distributions were developed to describe the water-to-milk transfer (Bq L^{-1} milk per Bq L^{-1} water) for ^{137}Cs, ^{90}Sr, ^{60}Co, and ^{131}I and the water-to-beef transfer (Bq kg^{1} meat per Bq L^{-1} water) for ^{137}Cs, ^{106}Ru, ^{90}Sr, and ^{60}Co. These parameters (summarized in Tables 9.10 and 9.11) are used to estimate the radionuclide concentrations in the milk and meat of cows that used the Clinch river as a source of drinking water. The primary source of uncertainty in the transfer of radionuclides to milk was the intake-to-milk transfer coefficient (F_m); similarly, the primary source of uncertainty in the transfer of radionuclides to meat was the intake-to-meat transfer coefficient (F_f).

9.6 References

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10.0 EXTERNAL EXPOSURE TO CONTAMINATED SHORELINE SEDIMENT

10.1 Background

Since 1944, individuals may have been exposed to radiation from contaminants in shoreline sediment at various locations along the Clinch River. Jones Island, Grassy Creek/K-25, Kingston Steam Plant, and the City of Kingston have been identified as the locations of greatest concern. The primary source of radiation is radionuclides released into the Clinch River from White Oak Lake and then deposited along the banks of the river. Some radionuclides easily attach to suspended sediment particles transported by the Clinch River and are deposited along with the sediments. Sediment deposition within a channel can occur along its entire wetted perimeter. During times of low flow or low water levels, these sediments become part of the shoreline, and individuals standing on the shoreline would then be exposed to radiation emissions from radionuclides present in the sediment.

The Task 4 screening calculations (Section 3) identified several radionuclides released to the Clinch River that are important for the external exposure pathway: ^{137}Cs, ^{60}Co, ^{106}Ru, ^{95}Zr, ^{95}Nb, ^{90}Sr, and ^{144}Ce. These radionuclides emit both beta and gamma radiation (Appendix 10-A). For ^{137}Cs, ^{60}Co, ^{95}Zr, and ^{95}Nb, the dose from photon emissions dominates the total dose. For ^{106}Ru and ^{144}Ce, beta radiation has a sizeable contribution. Strontium-90 and its radioactive daughter are beta emitters only.

This section discusses the necessary dose coefficients for estimating the external dose to individuals exposed to contaminants in the shoreline at various locations along the Clinch River. Various sources of bias and uncertainty that influence the dose estimates are analyzed. Several uncertain correction factors are derived for application to the published external dose coefficients.

10.2 Exposure Scenario

An individual standing or walking on the banks of the Clinch River would be externally exposed to radiations emitted by radionuclides that have been deposited directly on the shoreline. Radionuclides that do not attach to particulate matter are not deposited in the sediments; instead they flow with the water. Limited exposure results from photons that are emitted directly from radionuclides in the surface water. However, the contributions to the external dose by photons emitted from radionuclides in the flowing water can be neglected both because the radionuclide concentrations in water are low and because the emitted radiation is attenuated by the water.

Besides exposure from sediments underfoot, the shoreline on the opposite bank of the river and sediments deposited in the submerged part of the channel could also contribute to an individual's total photon exposure. Radiation emitted by radionuclides trapped in the deep sediment located on the bottom of the river is attenuated by the overlying river water. In other words, river water reduces the probability that a photon will reach the surface and potentially expose someone. To evaluate the shielding power of water,

the water depth for which a monoenergetic photon flux[1] is attenuated by a given percent was estimated using the methodology presented in Chilton et al. (1984). Photons of energies up to 2 MeV are found to be attenuated by more than 99% when submerged under 1 m of water (Table 10.1). For photons of lower energies, even less water is required for the same level of attenuation. For the seven important radionuclides released into Clinch River, the highest energy photon of important nuclear yield[2] is the 1.332 MeV photon emitted by ^{60}Co. For this photon, the water depth necessary for a 99% reduction in flux is about 80 cm. Therefore, contributions to exposure from deep sediments submerged in water by more than 1 m can be considered negligible. Sediments covered by less than one meter of water can be considered, conservatively, as part of the shoreline.

Table 10.1 Depth of a layer of water that would produce an attenuation of a monoen- ergetic photon flux by a given percent (based on methodology and data from Chilton et al., 1984).

	Level of attenuation		
Energy [MeV]	**99%**	**90%**	**80%**
	Water Depth [cm]		
0.6	51.5	25.7	18.0
1	65.1	32.6	22.8
1.5	80.2	40.1	28.0
2	93.4	46.7	32.6

Measurements of the water depth are available for the Clinch River. The profiles of the river bottom indicate that exposure to contaminated sediments can occur on only one side of the river in each section ("reach") of the river that is of interest, because of the steep slope of the opposite river bank (e.g., Figures 10.1 and 10.2). Therefore, contributions from the opposite shoreline can also be treated as being of negligible importance.

[1]Monoenergetic photon flux (density) = a parallel beam of photons of a given energy that pass through a plane in one direction.

[2]Nuclear yield = mean number of photons emitted per nuclear transformation of a radionuclide.

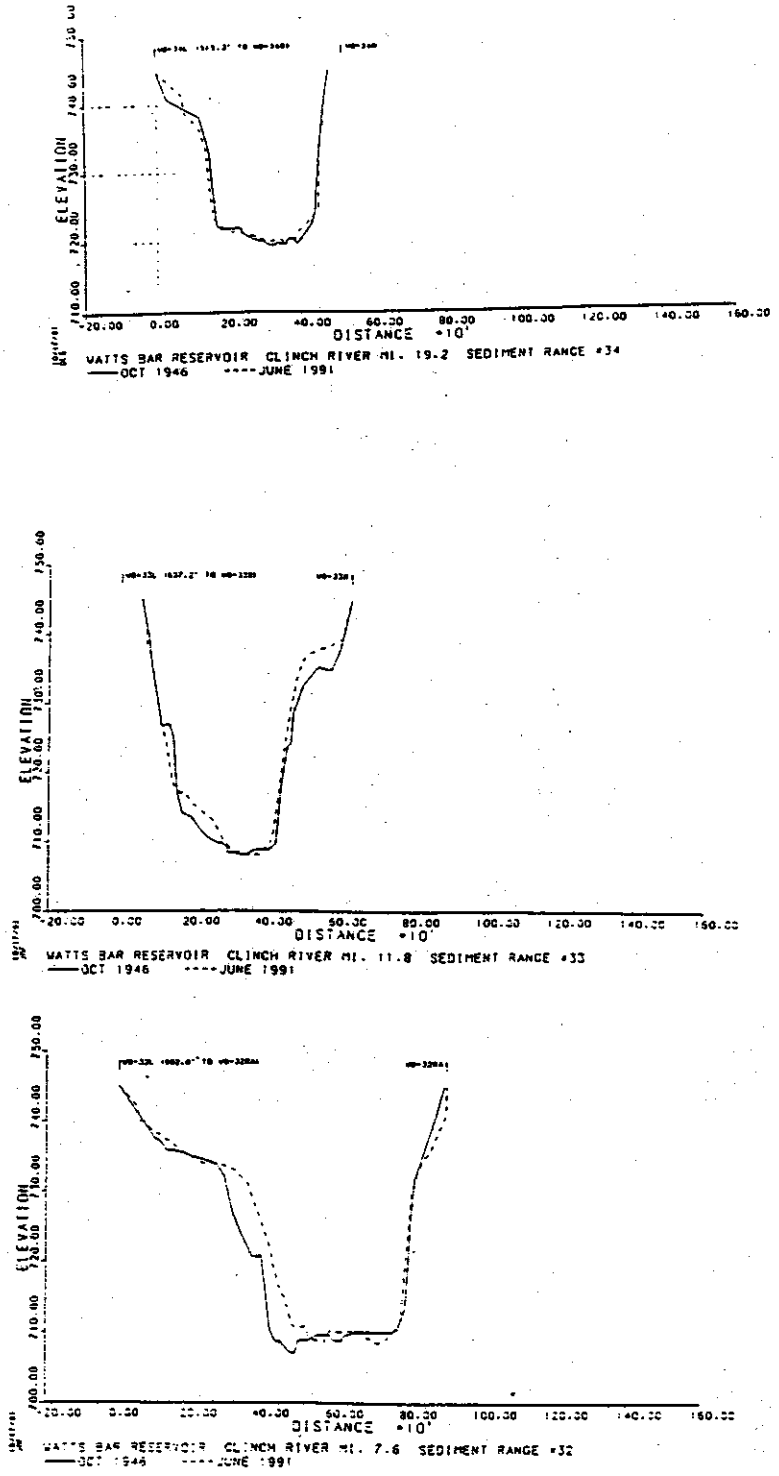

Figure 10.1 Profiles of the Clinch River bottom at CRM 19.2 (top), CRM 11.8 (middle), and CRM 7.6 (bottom).

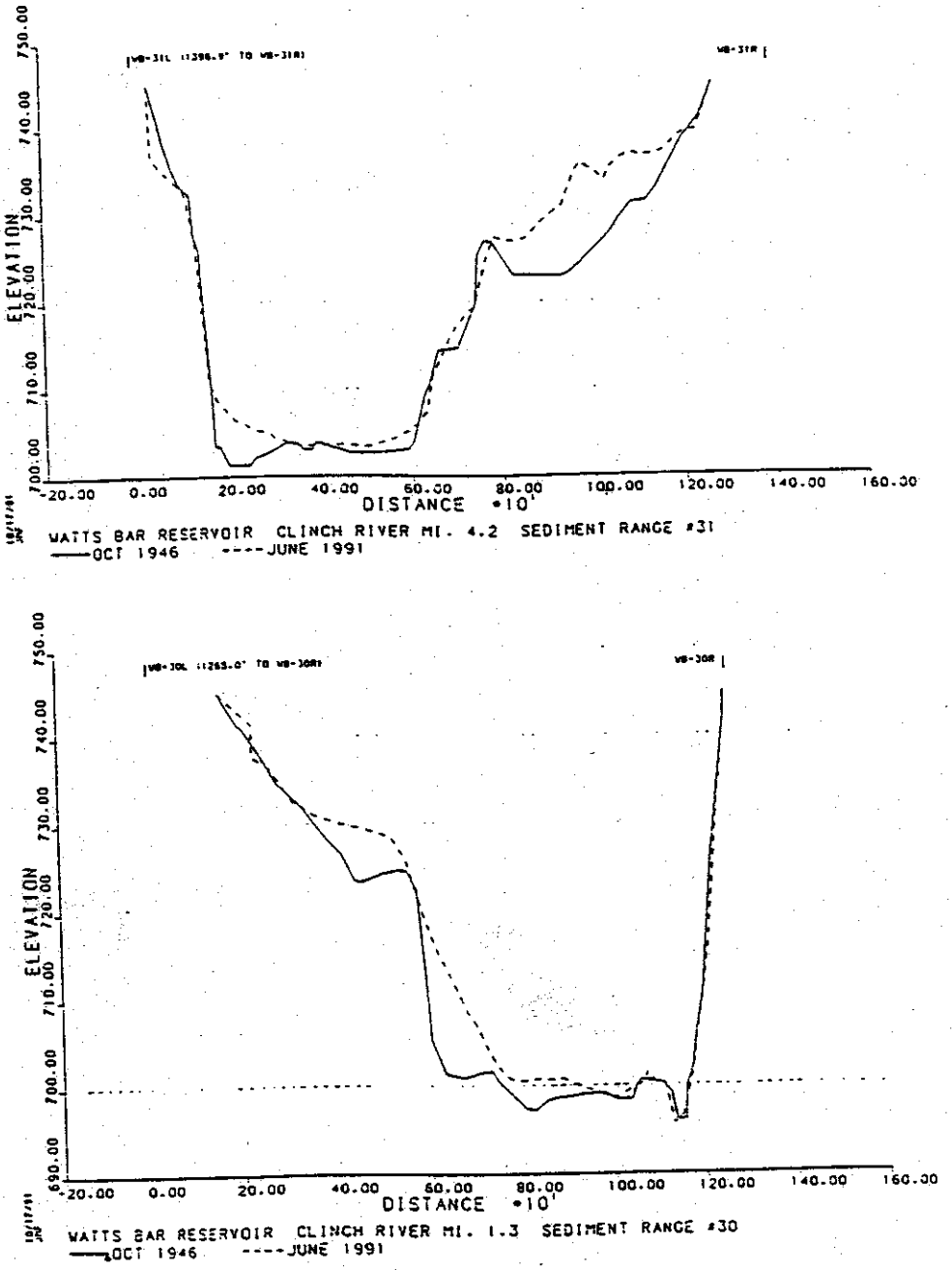

Figure 10.2 Profiles of the Clinch River bottom at CRM 4.2 (top), and CRM 1.3 (bottom).

The contaminated shoreline can be idealized as a strip of shoreline of finite width (including sediments under water up to a depth of 1 m) extending to an infinite length along the one side of the river. It is not necessary to consider the contribution from photons emitted from flowing surface water, deep sediments, or the opposite shoreline. For each reach of the river, the surface contamination of the shoreline is assumed to be uniform along the width and length of the shoreline. The width of the shoreline is subject to seasonal variations, which are explicitly considered in our approach (see Section 10.5 and Appendix 10.B). The contamination of the sediment is considered to vary from one year to another, as predicted by a water transport model (Section 6.0).

Details on the exposure frequency for the target individual spending time along the Clinch River shoreline are presented in Section 7. The mathematical formulation for estimation of risk from external exposure is presented in Section 4.

10.3 Modeling Approach

A generic approach to estimate the doses for people exposed to contaminated Clinch River shorelines is based on the concentration of the radionuclide in the contaminated sediment multiplied by a dose-rate factor.

$$D = \sum C_S \cdot DRF_{ext} \cdot EF \ \Delta t \qquad (10.1)$$

where

D = total dose to a given organ (Sv);
C_S = concentration of the radionuclide in the shoreline sediment (Bq kg^{-1});
DRF_{ext} = external dose-rate factor (Sv yr^{-1} per Bq kg^{-1}) defined as the dose received during the period of exposure by an individual standing on a shoreline having a unit sediment contamination (Equation 10.2);
EF = exposure frequency (unitless), and
Δt = 1 year and the summation is performed over the number of exposure years.

and

$$DRF_{ext} = DRF_{ext}^{published}(z) \cdot G \ H \qquad (10.2)$$

where

$DRF_{ext}^{published}(z)$ = are the published dose-rate factors (Eckerman and Leggett, 1996), expressed as a function of z
z = thickness of the contaminated sediment slab (m) (Section 10.4.1)
G = a geometry adjustment factor (unitless) that accounts for the particular geometry of the shoreline (Section 10.4.2), and

H = an uncertain correction factor (unitless) that accounts for other sources of uncertainty such as the non-homogeneity of the contamination across the shore and the movement of the individual on the contaminated surface (Section 10.4.4).

This is a generic equation included here for illustration purposes. The complete methodology is presented in Section 4. The dose-rate factors have been reported for exposure from an infinite plane or an infinite slab of contaminated ground surface (Kocher, 1983; Kocher and Sjoreen, 1985; Eckerman and Ryman, 1993; Eckerman and Leggett, 1996). In this study, the dose-rate factors $(DRF_{ext}^{published})$ published by Eckerman and Leggett (1996) were used. The exposure parameters (e.g., number of trips to the river per year, or number of hours per trip) are presented in Section 7, Table 7.5. The following section presents all other parameters necessary for dose estimation.

10.4 Parameters Necessary for External Dose Estimation

10.4.1 Thickness of the Contaminated Sediment Layer

Radionuclides have accumulated by sedimentation in the shorelines of Clinch River. The contamination is found in the top few centimeters of the sediment bed. Dose-rate factors are published (Eckerman and Leggett, 1996) as a function of the thickness (z) of a contaminated slab, for z = 1 cm, 5 cm, 15 cm, and for infinite depth (defined as four times the mean free path[3] of emitted photons.)

To calculate the dose-rate factor for any given thickness (z), an empirical observation was used. That is, it can be shown that the dose-rate factors varies with the thickness (z) of the contaminated slab as $[1-\beta*exp(-z/m_{soil})]$, where β is a constant, and m_{soil} is the mean-free-path of the radiation in soil. This property can be illustrated by plotting the dose-rate factors as a function of $exp(-z/m_{soil})$ (Figure 10.3).

Thus, to obtain the dose-rate factors for any given thickness, a linear fit to the curves shown in Figure 10.3 was used. This approximation is good for all internal organs, because the dose is dominated by gamma radiation emitted by the radionuclides (Figure 10.3a). Also, it holds for skin tissue (Figure 10.3b), for which the dose can be dominated by either gamma radiation (^{137}Cs, ^{60}Co, ^{95}Zr, ^{95}Nb), or by emitted electrons (^{90}Sr, ^{106}Ru, ^{144}Ce).

[3]Mean free path = the average length of the path of a particle from the point where it is emitted to the point where it has the next interaction, or the distance at which the radiation flux is decreased by e = 2.718.

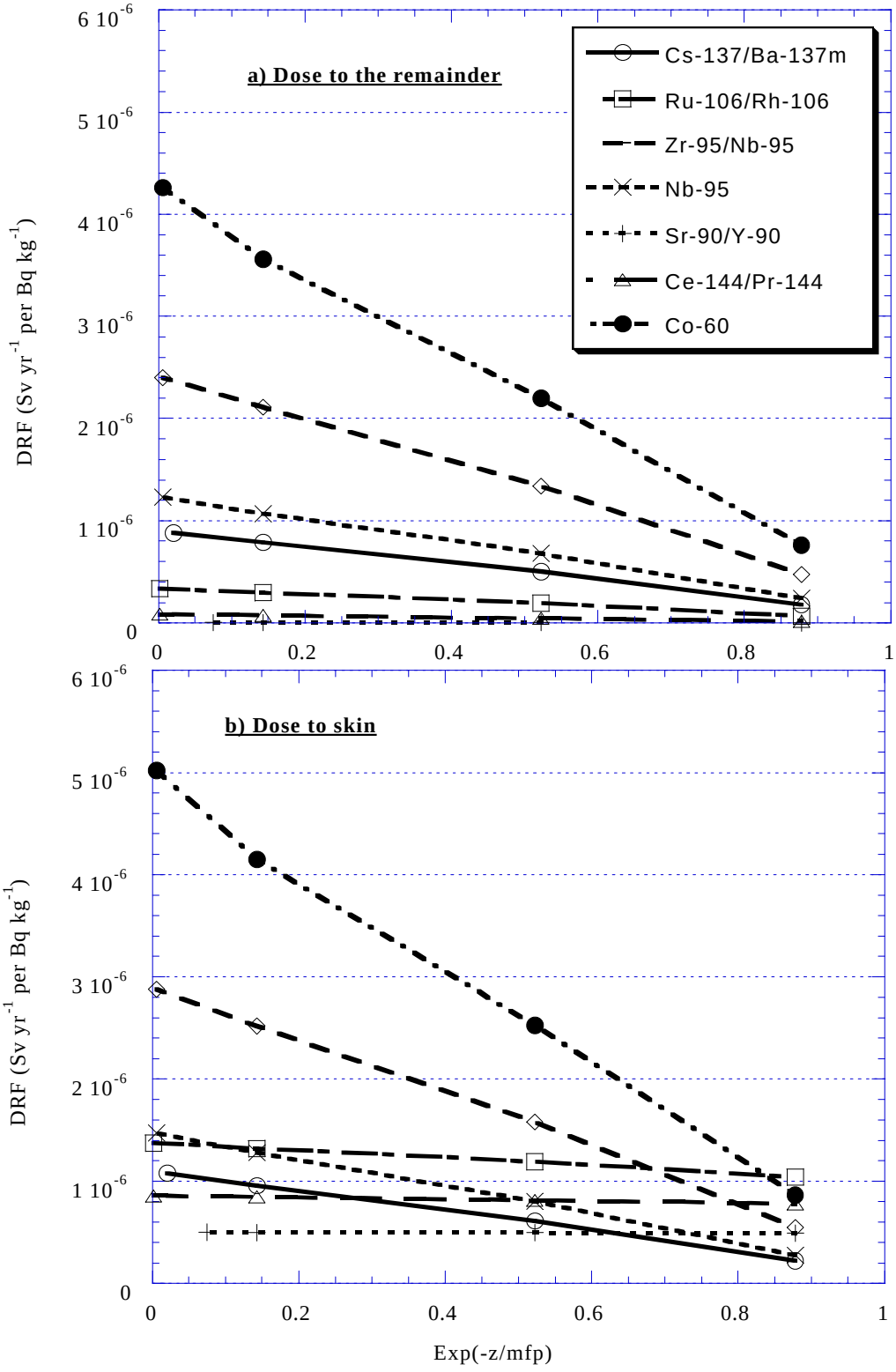

Figure 10.3 Dependence of the published external dose-rate factor (Eckerman and Leggett, 1996) on the depth (z) of an infinite uniform contaminated slab of soil expressed in terms of the mean free path (m_{soil}) of ^{137}Cs photons in soil.

The thickness of the contaminated top layer of the shoreline sediment depends on the sedimentation mechanisms. Brenkert (1996) indicated that the thickness of the contaminated sediment slab would be about 6 to 7 cm. In some cases, additional sediment from the hill-slopes of the shores may be deposited on the shoreline by run-off during rain. This uncontaminated sediment mixes with the contaminated sediment deposited by the water increasing the amount of sediment deposited. Brenkert (1996) also indicated that it is unlikely that the thickness of the contaminated slab would exceed 15 cm. On the other hand, during scouring events the sediment is removed faster than is deposited. We assumed that at least 2 cm of sediment is present at any time. The uncertainty in the thickness of the contaminated sediment was represented by a triangular distribution having a minimum of 2 cm, a most likely value of 7 cm and a maximum of 15 cm.

The thickness of the contaminated sediment layer was used as follows: for each sampled value of the thickness a dose-rate factor was estimated using the relationships plotted in Figure 10.3. This dose-rate factor is representative for an infinite slab of soil contaminated to the sampled depth. Then this dose-rate factor is further adjusted for a shoreline of a given width (Eq.10.2.)

The thickness of the contaminated sediment was applied for all locations and all years. In contrast, the width of the shoreline was derived for each location of interest and for all years (Appendix 10.B). The dose-rate factors derived by the above method for an area contaminated in the top 7 cm are shown in Table 10.2.

10.4.2 Geometry of Shoreline and Energy of the Gamma Radiation

A contaminated shoreline can be approximately described as a rectangular area. The dimension of this area parallel with the river and corresponding to shoreline length was found to be larger than three times the mean free path in air of the highest energy emitted by the contaminants (Table 10.3), and because of this, is considered infinite for radiation transport purposes. On the other hand, the dimension perpendicular to the river, corresponding to shoreline width, has a minimum length of about 2 meters and a maximum length that will not exceed distances comparable with the photon mean free path in air. To account for the geometry of a river shoreline, Eckerman and Ryman (1993) recommended usage of a dose-reduction factor of 0.2.

Table 10.2 Dose-rate factors (Sv yr^{-1} per Bq kg^{-1}) for exposure to soil contaminated to a depth of 7 cm (derived using the methodology presented in section 10.3.1 and based on data from Eckerman and Leggett, 1996).

Organ or Tissue	$^{137}Cs/^{137m}Ba$	$^{106}Ru/^{106}Rh$	$^{95}Zr/^{95}Nb$	^{95}Nb	$^{90}Sr/^{90}Y$	$^{144}Ce/^{144}Pr$	^{60}Co
Adrenals	2.53E-06	2.15E-07	1.44E-06	7.32E-07	3.23E-09	5.37E-08	2.53E-06
Bone Surface	3.87E-06	4.01E-07	2.47E-06	1.25E-06	8.70E-09	1.15E-07	3.87E-06
Brain	2.80E-06	2.46E-07	1.62E-06	8.25E-07	3.69E-09	6.03E-08	2.80E-06
Breast	3.04E-06	2.80E-07	1.82E-06	9.23E-07	4.50E-09	6.91E-08	3.04E-06
Digestive System							
Oral Cavity	2.81E-06	2.50E-07	1.64E-06	8.38E-07	3.91E-09	6.18E-08	2.81E-06
Esophagus	2.38E-06	2.06E-07	1.36E-06	6.92E-07	3.03E-09	5.07E-08	2.38E-06
Stomach	2.56E-06	2.24E-07	1.49E-06	7.57E-07	3.48E-09	5.59E-08	2.56E-06
Small Intestine	2.44E-06	2.09E-07	1.39E-06	7.09E-07	3.15E-09	5.21E-08	2.44E-06
Upper Larger Intestine	2.48E-06	2.15E-07	1.42E-06	7.26E-07	3.26E-09	5.33E-08	2.48E-06
Colon	2.49E-06	2.17E-07	1.43E-06	7.28E-07	3.26E-09	5.35E-08	2.49E-06
Rectum	2.49E-06	2.17E-07	1.43E-06	7.28E-07	3.26E-09	5.35E-08	2.49E-06
Gallbladder	2.49E-06	2.08E-07	1.42E-06	7.26E-07	3.16E-09	5.19E-08	2.49E-06
Heart	2.54E-06	2.22E-07	1.47E-06	7.51E-07	3.43E-09	5.52E-08	2.54E-06
Kidneys	2.59E-06	2.31E-07	1.51E-06	7.69E-07	3.58E-09	5.74E-08	2.59E-06
Liver	2.59E-06	2.29E-07	1.50E-06	7.67E-07	3.55E-09	5.68E-08	2.59E-06
Lungs	2.79E-06	2.49E-07	1.63E-06	8.32E-07	3.88E-09	6.17E-08	2.79E-06
Muscle	2.83E-06	2.53E-07	1.66E-06	8.45E-07	3.95E-09	6.24E-08	2.83E-06
Ovaries	2.47E-06	2.07E-07	1.39E-06	7.11E-07	3.11E-09	5.17E-08	2.47E-06
Pancreas	2.36E-06	2.04E-07	1.34E-06	6.85E-07	3.08E-09	5.11E-08	2.36E-06
Prostate	2.81E-06	2.50E-07	1.64E-06	8.38E-07	3.91E-09	6.18E-08	2.81E-06
Red Bone Marrow	2.80E-06	2.46E-07	1.62E-06	8.28E-07	3.70E-09	6.03E-08	2.80E-06

Table 10.2 (continued)

Organ or Tissue	$^{137}Cs/^{137m}Ba$	$^{106}Ru/^{106}Rh$	$^{95}Zr/^{95}Nb$	^{95}Nb	$^{90}Sr/^{90}Y$	$^{144}Ce/^{144}Pr$	^{60}Co
Skin	3.28E-06	1.26E-06	1.96E-06	9.96E-07	4.98E-07	8.31E-07	3.28E-06
Spleen	2.60E-06	2.30E-07	1.51E-06	7.71E-07	3.57E-09	5.71E-08	2.60E-06
Testes	2.99E-06	2.73E-07	1.77E-06	9.02E-07	4.34E-09	6.77E-08	2.99E-06
Thymus	2.69E-06	2.39E-07	1.58E-06	8.04E-07	3.70E-09	5.88E-08	2.69E-06
Thyroid	2.64E-06	2.34E-07	1.54E-06	7.83E-07	3.69E-09	5.86E-08	2.64E-06
Urinary Bladder	2.56E-06	2.21E-07	1.47E-06	7.48E-07	3.42E-09	5.57E-08	2.56E-06
Uterus	2.42E-06	2.07E-07	1.39E-06	7.07E-07	3.11E-09	5.14E-08	2.42E-06
Remainder[a]	2.81E-06	2.50E-07	1.64E-06	8.38E-07	3.91E-09	6.18E-08	2.81E-06
Effective (ICRP)[b]	2.75E-06	2.55E-07	1.61E-06	8.18E-07	8.74E-09	6.85E-08	2.75E-06

[a] The values for remainder are calculated as a function of the doses for the nine organs or tissues for which ICRP (1990) doses not assign a tissue weighting factor.

[b] The effective dose is the average of the doses to each organ weighted by ICRP (1990) tissue weighting factors.

Table 10.3 Mean free path of gamma radiation and the empirically derived fit coefficients (*a* and *b*) for the Berger formulation for the radiation buildup factor (Chilton et al., 1984).

Energy [MeV]	**mean free path in air (1/μ[a]) [m]**	**Coefficient for the buildup factor**	
		a	***b***
0.1	56.1	5.93	0.113
0.661[c]	103.4	1.815	0.049
0.75[d]	113.8	1.795	0.0483
1.252[e]	144.2	1.38	0.028
3	232.7	0.75	0.005

[a] F = attenuation coefficient (Eq. 10.3).
[b] the buildup factor is defined using Berger Formulation (Eq. 10.5).
[c] energy of the gamma radiation emitted by $^{137}Cs/^{137}Ba$.
[d] energy representative of the gamma radiation emitted by $^{95}Zr/^{95}Nb$.
[e] average energy of the gamma radiation emitted by ^{60}Co.

In this study, a geometry adjustment factor (*G*) accounting for the finite shoreline geometry is defined as the ratio of the dose to an individual located in the center of the rectangular surface, to the dose to the same individual from an infinite surface, when the surfaces are contaminated at the same concentration. For a photon of a given energy, this factor can be approximated by the ratio of the "free-field"[4] gamma ray fluxes:

$$G = \frac{Dose_{finite\&surface}}{Dose_{infinite\&surface}} = \frac{Flux_{finite\&surface}}{Flux_{infinite\&surface}} \qquad (10.3)$$

where

Flux = the free-field flux from a finite and an infinite surface, respectively (cm^{-2}).

The fluxes are calculated at a point 1 meter above ground in a gamma-ray "free-field." The contamination of the surface is assumed to be uniform (S_A, Bq m^{-2}). Attenuation in air and "buildup"[5] due to the photons scattered by air are considered. Similar methodologies for estimation of fluxes have been employed by Kocher (1983), Kocher and Sjoreen (1985), and Chilton et al. (1984).

[4] "Free-Field" flux = the flux of radiation unperturbed by the presence of the human body.
[5] Buildup = an increase in the flux of photons at a given location because of the contributions of the photons scattered by the air.

For a given photon energy, the differential flux at point P is given by

$$dF \propto \frac{B(\mu r)\cdot e^{-\mu r}}{4\pi r^2}\cdot S_A dA \qquad (10.4)$$

where

dF = differential flux (Bq m^{-2});
$B(\mu r)$ = empirically derived buildup factor (unitless);
$S_A\, dA$ = differential plane source term, considered as point source (Bq);
μ = attenuation coefficient (m^{-1}; Table 10.3); and
r = the distance from the differential source to the point of interest P (m).

$r = \sqrt{x^2 + y^2 + h^2}$; x, y = the Cartesian coordinates of a point source

and h = 1 m above the ground surface.

The Berger formulation (Eq. 10.5) was used to describe the buildup factor.

$$B(\mu r) = 1 + a\cdot\mu\cdot r\cdot e^{b\cdot\mu\cdot r} \qquad (10.5)$$

where a and b are empirically derived, unitless parameters which depend on the photon energy (Table 10.3; Chilton et al., 1984).

Finally, the flux is computed as follows.

$$Flux = 4\cdot\int_0^{\frac{w}{2}}\int_0^{\infty} dF\cdot dx\cdot dy \qquad (10.6)$$

where w is the width of the shoreline area, and the factor of four arises because the integral is over one-fourth of the shoreline. For the rectangular area, the integration was performed numerically in the Cartesian coordinate system. For the infinite plane, an analytical expression of the flux was used.

The geometry factors were derived for a number of photon energies (Table 10.4) and for shoreline widths ranging from 2 meters up to 100 meters (Table 10.5, Figure 10.4).

Table 10.4 Selected nuclear properties of the radionuclides of interest for the external exposure pathway for the Clinch River.

Radionuclide	**Photon energy range[a] [MeV]**		**Representative energy [MeV]**
	minimum	**maximum**	
$^{137}Cs/^{137m}Ba$	0.661	0.661	0.661
^{60}Co	1.173	1.332	1.25
$^{106}Ru/^{106}Rh$	0.428	2.406	0.661
$^{95}Zr/^{95}Nb$	0.724	0.766	0.75
$^{144}Ce/^{144}Pr$	0.033	2.186	0.1 (for ^{144}Ce) and 0.661 (for ^{144}Pr)[b]

[a] based on radiation tabulated in ICRP (1983); see Appendix 10-A.
[b] see discussion.

Table 10.5 Geometry factors to be applied to the dose-rate factor for infinite extent as a function of shoreline width, for different gamma energies.

Width (m)	**Radionuclide and Representative Energy (MeV)**					
		$^{144}Ce/^{144}Pr$	**^{137}Cs, ^{90}Sr, $^{106}Ru/^{106}Rh$**	**$^{95}Zr/^{95}Nb$**	**^{60}Co**	
	0.1 MeV	-- [a]	0.661 MeV	0.75 MeV	1.252 MeV	3.0 MeV
2	0.11	0.14	0.15	0.15	0.15	0.16
4	0.19	0.22	0.24	0.25	0.25	0.25
5	0.21	0.26	0.28	0.28	0.29	0.29
10	0.32	0.37	0.39	0.39	0.4	0.41
15	0.39	0.44	0.46	0.46	0.47	0.47
20	0.45	0.49	0.51	0.51	0.52	0.52
30	0.53	0.56	0.58	0.58	0.59	0.59
50	0.65	0.66	0.67	0.67	0.68	0.67
75	0.74	0.74	0.74	0.75	0.75	0.74
100	0.8	0.79	0.79	0.79	0.79	0.78

[a] 0.1 MeV (for ^{144}Ce) and 0.661 MeV (for ^{144}Pr); see text for details.

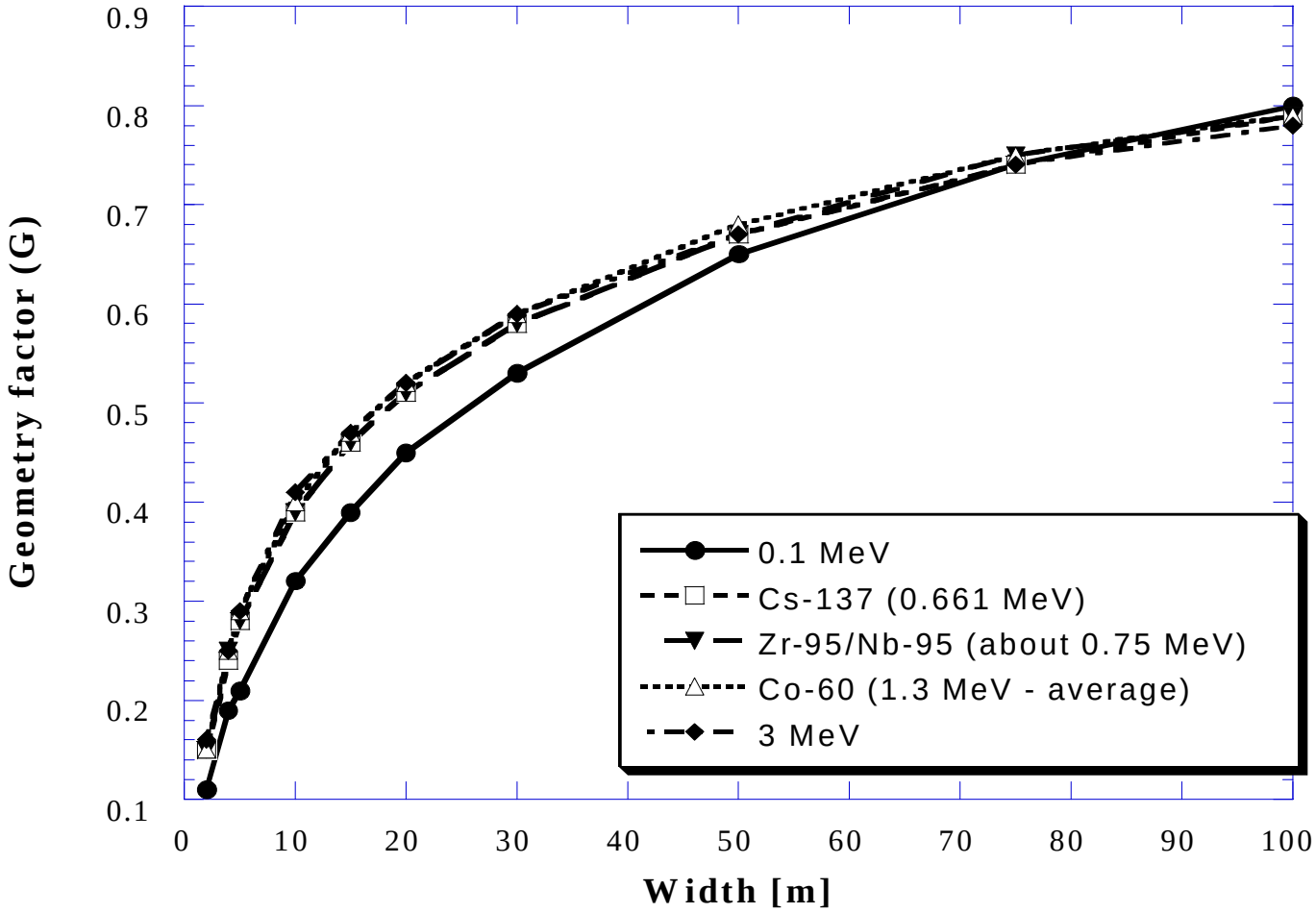

Figure 10.4 The geometry factor as a function of width of the shoreline for selected gamma energies.

The lowest and the highest chosen energies represent a lower and an upper limit for the energy of the gamma rays emitted by most of radionuclides of interest (Table 10.4). As shown graphically in Figure 10.4, the geometry factor is rather insensitive to photon energy.

For $^{137}Cs/^{137m}Ba$, the only gamma emission (0.661 MeV) was selected as the representative energy for estimating the geometry factor. Cobalt-60 emits two photons of similar energies (1.173 MeV and 1.332 MeV) with essentially the same nuclear yield. Given the insensitivity of the geometry correction factor to small variations in energy (Figure 10.5), the average energy of the two photons (1.252 MeV) was chosen as the representative energy for ^{60}Co. Three gamma rays of practically the same energies (0.72, 0.76, 0.77 MeV) are emitted by ^{95}Zr and its daughter ^{95}Nb. A geometry factor was estimated for these radionuclides, using a representative gamma energy of 0.75 MeV.

For a given shoreline width, the geometry factors are essentially independent of photon energy at energies larger than 0.6 MeV (Figure 10.5). For lower energy photons, however, a significant difference was found. For example, the geometry factors for 0.1 MeV photons are about 15% - 25% lower than the factors for photons of 1.3 MeV for shoreline widths of 5 to 50 m. This difference decreases with increasing shoreline widths (Figure 10.5).

Ruthenium-106 and its daughter ^{106}Rh emit a large number of photons, which have energies ranging from 0.428 MeV to 2.406 MeV (Table 10.4). The first gamma ray with a large nuclear yield has an energy of 0.512 MeV. For this range of energies, the geometry factors remain approximately the same for a given shoreline width (Figure 10.4). Therefore, the correction factor for any of the intermediate energies can be assumed to be a good representation of the real correction factor for $^{106}Ru/^{106}Rh$. In this assessment, the geometry factors obtained for $^{137}Cs/^{137}Ba$ were also applied to $^{106}Ru/^{106}Rh$.

A geometry factor for $^{144}Ce/^{144}Pr$ is more difficult to assess, due to the large spectrum of energies emitted by these isotopes. Cerium-144 emits photons in a range from 0.033 MeV to 0.133 MeV (ICRP, 1983). The most probable photon emission (10.8%) is the most energetic one, 0.133 MeV. The x-ray emissions have an energy varying from 0.005 MeV to 0.036 MeV. Praseodymium-144, the daughter of ^{144}Ce, emits a photon of 0.697 MeV energy in 1.5% of its decays. The other listed photon emissions (ICRP, 1983) have energies up to 2.186 MeV, but very low probabilities of occurrence. For most organs, the contribution to the external dose from ^{144}Pr is about twice as high as the contribution from ^{144}Ce. A representative reduction factor for ^{144}Ce (emitting low energy photons) can be obtained by using the energy of 0.1 MeV (Table 10.5). On the other hand, for ^{144}Pr the reduction factor obtained for 0.661 MeV is appropriate. Therefore, a reduction factor for $^{144}Ce/^{144}Pr$ (Table 10.5) was determined as a weighted combination of the reduction factors for ^{144}Ce (one-third) and for ^{144}Pr (two-thirds).

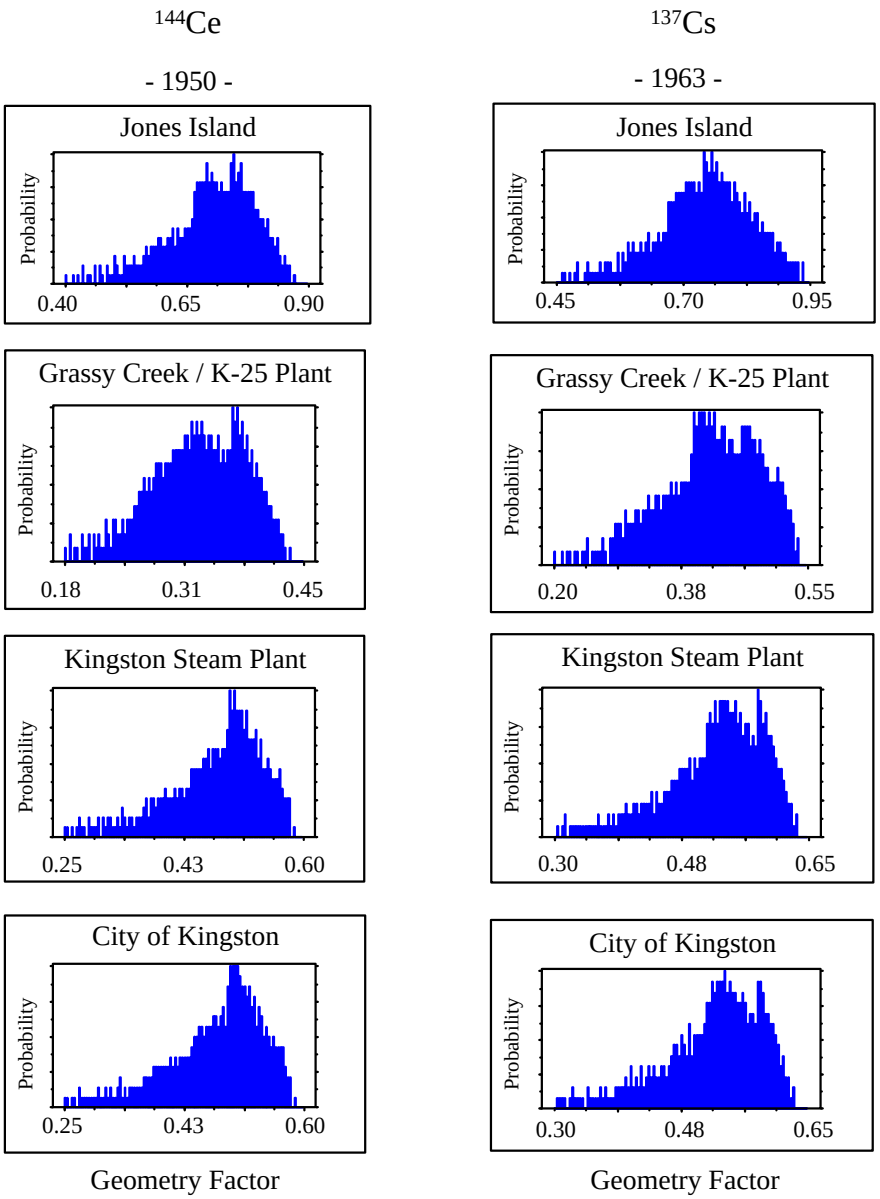

Figure 10.5 Examples of probability distributions describing the geometry factor for exposure to contaminated shoreline at various reaches of the Clinch River, for two radionuclides and two years. The calculated geometry factor is shown on the horizontal axis, while the vertical axis represents the relative probability of each value.

The organ dose rates for $^{90}Sr/^{90}Y$ (which are beta emitters) are due to photons arising as the emitted beta particles lose their kinetic energy within the ground and air. This process is called bremsstralung. Only a small fraction of the kinetic energy of the beta radiations is converted to photons; as a result the numerical value of the dose-rate factor for these radionuclides is one to two orders of magnitude smaller than for other radionuclides. The energy of the bremsstralung radiation produced by $^{90}Sr/^{90}Y$ ranges from zero to the maximum energy of the emitted beta radiation (about 2.3 MeV). Since the bremsstrahlung energy spectrum was not available for $^{90}Sr/^{90}Y$, a geometry factor for a 0.661 MeV energy was selected. This choice is conservative, because the geometry factors are lower for lower energies, when more attenuation is produced.

The above approach and discussions apply for all organs other than skin. For external irradiation from contaminated area sources, the dose is given by the contribution of both electrons and gamma rays emitted by each radionuclide. For all internal organs, the dose is dominated by photons, because electrons are stopped shortly after penetrating through the skin. For skin, however, electrons can dominate the dose from a given radionuclide. This is the case for $^{90}Sr/^{90}Y$, for $^{106}Ru/^{106}Rh$, and for $^{144}Ce/^{144}Pr$ (Table 10.6). Transport of electrons through air, from the contaminated ground to the body surface, is different from the transport of gamma radiation. Electrons travel much shorter distances in air than do photons. Practically, a small contaminated area can produce the same electron dose to the skin as a very large (infinite) contaminated area. Thus, for $^{90}Sr/^{90}Y$, $^{106}Ru/^{106}Rh$, and $^{144}Ce/^{144}Pr$ and for skin tissue only, no adjustment for the geometry of the shoreline is necessary; that is, for these radionuclides, the doses to skin should be calculated based solely on the dose-rate factors for an infinite area. For ^{137}Cs, ^{60}Co, and $^{95}Zr/^{95}Nb$, the dose-rate factors for skin are adjusted using the calculated geometry factor as for all other organs. Based on sample calculations, the total dose to the skin from external exposure in this study is expected to be about 15-20% higher if the geometry factor is omitted for $^{90}Sr/^{90}Y$, $^{106}Ru/^{106}Rh$, and $^{144}Ce/^{144}Pr$. However, because these isotopes do not contribute significantly to the total dose and risk from external exposure (Section 13), this correction was not made. In the interest of calculational simplicity, the geometry factors were used for all radionuclides and all organs.

Table 10.6 A breakdown of the contribution of electrons and gamma radiation to the dose-rate factors to skin from exposure to contaminated ground surfaces for the radionuclides of interest. Values are given in Sv s^{-1} per Bq m^{-2}.

Radionuclide	Total dose-rate factors[a]	Electron dose-rate factors[b]	Gamma dose-rate factors[c]	Dominant contribution
^{90}Sr	1.40E-16	4.12E-17	9.88E-17	gamma
^{90}Y	1.05E-14	7.29E-15	3.21E-15	electron
^{90}Sr/^{90}Y	1.06E-14	7.33E-15	3.31E-15	**electron**
^{137}Cs	2.75E-16	8.88E-17	1.86E-16	gamma
^{137m}Ba	1.65E-15	2.73E-16	1.38E-15	gamma
^{137}Cs/^{137m}Ba	1.93E-15	3.61E-16	1.56E-15	**gamma**
^{106}Ru	0.00	0.00	0.00	none
^{106}Rh	1.40E-14	1.05E-14	3.54E-15	electron
^{106}Ru/^{106}Rh	1.40E-14	1.05E-14	3.54E-15	**electron**
^{95}Zr	8.90E-16	6.98E-18	8.83E-16	gamma
^{95}Nb	9.05E-16	6.98E-18	8.98E-16	gamma
^{95}Zr/^{95}Nb	1.80E-15	1.40E-17	1.78E-15	**gamma**
^{144}Ce	2.61E-17[d]	Not reported	2.61E-17[d]	gamma
^{144}Pr	1.27E-14[e]	Not reported		electron
^{144}Ce/^{144}Pr	1.27E-14			**electron**
^{60}Co	2.76E-15[f]	Not reported	2.76E-15[f]	**gamma**

[a] From Eckerman and Ryman (1993).
[b] From Kocher and Eckerman (1981).
[c] Obtained by subtracting second column from the first column.
[d] Total dose rate to skin is only marginally higher than the dose rate to the breast tissue or the dose rate to the gonads.
[e] Total dose rate to skin is three orders of magnitude larger than to any other organs.
[f] Total dose rate to skin is practically equal to the dose rate to any other organ.

The geometry factor was estimated as a function of the width of the rectangular area describing the shoreline (Figure 10.4). The shoreline width varies from one season to another during a given year, from one year to another, and from one location to another (Appendix 10.B). To express the uncertainty introduced by the annual variation in the shoreline width the following procedure was used.

(a) The maximum and the average shoreline width in a given year were estimated according to the water transport model (HEC-6-R) output and the bathymetric measurements available for the Clinch River (Appendix 10.B).

(b) A minimum shoreline width of 2 meters was assumed for all years,

(c) The uncertainty in the size of the shoreline during a year was expressed as a triangular distribution with a minimum of 2 m, a mode equal to the estimated average shoreline width in each year and a maximum given by the estimated maximum shoreline width in the same year.

(d) The probability distributions for the shoreline width were sampled independently for each year and each location.

(e) A geometry factor was obtained using the relationship in Figure 10.4; linear interpolation was used to obtain intermediate points.

The result of this procedure is a set of probability distributions that describe the uncertainty introduced by the geometry of the Clinch River shoreline. There is one for each exposure year (48 years), for each radionuclide of interest (7 radionuclides), and for each location of interest (4 locations). These distributions are applied to the dose-rate factors for each specific organ. An example of the distributions for such geometry factors is presented in Figure 10.5. All the derived geometry factors are presented in Appendix 10.B.

10.4.3 Organ Masses

The dose-rate factors have been calculated for a reference individual (Eckerman and Ryman, 1993; Eckerman and Leggett, 1996). Differences between the prescribed values of organ mass for the reference individual and those for real individuals exposed along the Clinch River can arise from natural variability and from sex-specific factors. The uncertainties in the specifications of the target organ mass were taken into account as follows for the estimation of total dose to each organ from all routes of exposure:

(a) The uncertainties in the dose-rate factors were calculated by applying the geometry factors to the published dose-rate factors.

(b) The probability distributions for dose-rate factors for external exposure for a given organ were multiplied by the mass of that organ as specified in the ICRP methodology; the quantity obtained represents the probability density function for energy deposition rate per unit of surface contamination for that particular organ.

(c) The energy deposited by external exposure to a given organ was added to the energy deposited in the same organ by all other radionuclides and exposure pathways.

(d) The total energy deposition was then divided by the probability distribution that expresses the uncertainty in the organ mass, to obtain the total dose to that organ.

The method outlined here, together with the probability distributions for all organ masses, is presented in more detail in Section 11.1.

10.4.4 Other Sources of Uncertainty

The dose rate conversion factors are estimated on the basis of the assumption that the ground surface is contaminated uniformly (or homogeneously). The shorelines are contaminated by the deposition of the sediment carried by the river. Thus, the concentration in the shoreline sediment could change from the water line to the far end of the shore. Since heterogeneity of shoreline contaminant concentrations would largely be the result of variations in deposition and scouring patterns on a daily scale, the assumption of homogeneity on an annual time scale is a reasonable assumption. However, there is uncertainty in the knowledge of the exact value of the average (homogeneous) shoreline concentration.

The dose rate factors are estimated on the assumption that the individual is located in the center of the contaminated shoreline. In reality, people spend time on the Clinch River shoreline walking or playing. Thus another source of uncertainty will be introduced by the movement of the individual on the shoreline.

These sources of uncertainty are small, but hard to quantify. A factor (H), having a uniform distribution between 0.95 and 1.05, is used to account for these sources of uncertainty.

10.5 Summary

This section presents details of the approach used for estimating external doses to people exposed to radionuclides accumulated on the shorelines of the Clinch River. Jones Island, Grassy Creek/K-25, Kingston Steam Plant, and the City of Kingston have been identified as the locations of greatest concern. The screening calculations (Section 3) identified seven radionuclides released to the Clinch River from the Oak Ridge National Laboratory that are important for the external exposure pathway: ^{137}Cs, ^{60}Co, ^{106}Ru, ^{95}Zr, ^{95}Nb, ^{90}Sr, ^{144}Ce.

The main source of external exposure to Clinch River is the underfoot shoreline sediment. The radionuclides in the sediments on the bottom of the river and in the river water contribute little to external exposure, and their contribution can be neglected. Also, the profiles of the river bottom show that, in most cases, only one shore of the river is accessible for people, while the opposite shore is steep. Thus, exposure could occur only on one side of the river. Based on these observations, the shoreline was idealized as a strip of land having an infinite length and a finite width (as compared to the distance traveled by radiation in air.)

Radionuclides accumulated by sedimentation are found in the top few centimeters of the sediment bed. The thickness of the contaminated layer is about 6-7 cm and it can vary between 2 and 15 cm. The shoreline width for each location of interest was obtained from bathymetric measurements of the Clinch River (Appendix 10.B) for all years of interest. The minimum shoreline is considered to be at least 2 m. The maximum estimated shoreline width is 34 m in the Jones Island area, 25 m in the Grassy Creek/K-25 area, 49 m at Kingston Steam Plant, and 48 m in the City of Kingston.

To estimate the doses from external exposure, dose-rate factors based on the values reported by the International Commission on Radiological Protection (ICRP) were used. First, dose-rate factors were estimated as a function of the thickness of the contaminated layer (Section 10.4.1). Then, they were adjusted for the finite width of the Clinch River shorelines (Section 10.4.2, Appendix 10.B). Finally, external doses are obtained by combining the concentrations of the radionuclide in sediment (Section 6) with the dose-rate factors (Section 10.3) and with the exposure parameters (Section 7).

10.6 References

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11.0 INTERNAL DOSIMETRY

Out of the large number of radionuclides released into the Clinch River from White Oak Creek, five could potentially affect individuals consuming fish, beef, milk, or drinking water: ^{137}Cs, ^{90}Sr, ^{60}Co, ^{106}Ru, and ^{131}I (Section 3). To determine possible effects, the internal dosimetry methodology from the International Commission on Radiological Protection (ICRP) was used to calculate doses to people consuming contaminated food items or drinking water. The purpose of this section is to calculate the ingestion dose factors[1] for the radionuclides of interest and to determine the uncertainties associated with these factors for different target individuals. The uncertainties are reported using a 95% confidence interval (defined as the range within which the true but unknown value is contained with 95% confidence), and using "a factor of *n*" (defined as the ratio between the upper limit of the confidence range and the median value).

Internal dosimetry describes the fate of an ingested radionuclide in the body, starting with the passage through the gastrointestinal tract; continuing with absorption to blood, distribution of the radionuclide to various organs in the body, and retention of the radionuclide in these organs; and ending with its elimination by radioactive decay and excretion. Radiation emitted during the radionuclide's transport through the body deposits energy, which delivers a radiation dose to individual organs. The methodology for estimating the ingestion dose factors is described in Section 11.1. The first sections describe the three parts of the methodology that apply to all radionuclides: (a) the movement through the gastrointestinal tract (Section 11.1.1), (b) the fraction of the energy emitted in an organ that is absorbed in other organs (Section 11.1.2), and (c) the organ masses (Section 11.1.3). Radionuclide- specific information such as the absorption from the gastrointestinal tract to blood and metabolism of the radionuclide in the body is described in Sections 11.2 to 11.6.

The target individuals are defined as adults ingesting ^{137}Cs, ^{90}Sr, ^{60}Co, or ^{106}Ru. Differences between genders are accounted for in terms of gender-specific organs, but not in the biokinetics and metabolism of each individual element. The critical population group for exposure to ^{131}I is children under 15 years of age, because of the preferential accumulation of ^{131}I in the thyroid gland and because of the greater risk of radiation-induced thyroid cancer in children. Therefore, the target individual for internal dosimetry of ^{131}I is a child at various ages up to age 15.

The internal dosimetry methodology was based on the ICRP models for the transfer and bioaccumulation of the radionuclides in the body. To obtain estimates of the ingestion dose factors and their uncertainty, different approaches were used, according to the amount and quality of the available data. The ICRP ingestion dose factors for ^{137}Cs, ^{60}Co, and ^{106}Ru were modified by multiplicative uncertainty factors. Probability distribution functions expressing the uncertainty of the biokinetic parameters were defined and propagated through the ICRP models for ^{137}Cs, ^{106}Ru, ^{90}Sr and ^{131}I. For ^{60}Co, however, the uncertainty factor was developed based on the information on long-term retention of this element in different human subjects. New ingestion dose factors and associated uncertainties were calculated for ^{90}Sr and ^{131}I. ICRP

[1] An *ingestion dose (conversion) factor* is defined as the committed dose from ingestion of a unit of radioactivity (Sv Bq^{-1}). They are defined for various organs of the human body and for individual radionuclides.

Publication 67 (1993) reports dose factors for 22 different organs and an effective dose factor calculated as a weighted average of the dose factors for each organ. The weights for each organ are called "tissue weighting factors" and are determined by ICRP (1991) for 12 organs based on radiogenic cancer mortality data. A common weight is given for the remainder organs[2]. ICRP also reports a special dose factor formulated in terms of the average dose for the tissue comprising the remainder.

For the purpose of estimating the total risk of radiogenic cancer in this assessment, the dose factors were calculated for five additional organs that are important cancer sites: oral cavity, esophagus, rectum, gallbladder, and prostate. Colon and rectum are the last parts of the large intestine, and they are similarly irradiated by an ingested radioactive substance. Thus, the dose factor estimated for the colon was assigned also to the rectum. The ICRP dose factor assigned for the remainder tissues was used for the oral cavity, esophagus, gallbladder, and prostate. The method for estimating the total risk from ingested radionuclides is described in Section 12.2.6.

Uncertainties in the dose factors were estimated for all 27 organs. Together with the uncertainty analysis, a sensitivity analysis was performed to identify the most important contributors to the uncertainty in the ingestion dose factors.

11.1 Internal Dosimetry Methodology

A radionuclide enters the gastrointestinal tract when contaminated food or water is ingested. From the stomach, the radionuclides in combination with the stomach contents are transferred to the small intestine, where most of the nutrients are absorbed by the body. Radionuclides often mimic certain nutrients required by the body (e.g., ^{137}Cs behaves like potassium, and ^{90}Sr behaves like calcium), and therefore the radionuclides are absorbed into the bloodstream. Once in the bloodstream, they are deposited in other organs or excreted. Those radionuclides in the small intestine that are not absorbed by the bloodstream are transported from the small intestine to the colon and finally excreted. From a specific organ, the radionuclides may be transferred back to the blood or plasma prior to elimination from the body after filtration by the kidneys, secretion into the gastrointestinal tract, or entry into generally minor excretion pathways (e.g., sweat) (see Section 11.3 on ^{90}Sr).

The radionuclide accumulated in a given organ (usually called the “source” organ) undergoes radioactive decay, irradiating the surrounding organs (called “target” organs). A substantial portion of the emitted radiation will be deposited in the source organ itself. The radionuclides accumulated in every source organ of the body contribute to the dose to a given target organ. Some contributions might be negligible due to the relative position of the source-target pair or the type of radiation emitted by the radionuclide (e.g., ^{90}Sr emits beta radiation which is considered nonpenetrating).

[2]The “remainder” includes all organs or tissues for which no specific weighting factor is available (including most of the mass of the body, especially the muscles and much of the bone mass). The remainder as used by ICRP and by SENES for this assessment are similar but not entirely identical.

For adults, the dose to a given target organ (T) from a unit intake of radionuclide (the dose factor, Sv Bq^{-1}) is defined as follows:

$$DF_T = \sum_S U_{nt,s} \cdot SEE(T7S) \qquad \text{(Eq. 11.1)}$$

where

SEE (T 7 S)	=	the Specific Effective Energy (SEE) deposited in the target organ (*T*) from the radionuclide(s) accumulated in the source organs (*S*) (Sv per nuclear transformation*),
$U_{nt,S}$	=	the number of nuclear transformations produced by the radionuclide in the source organ from a unit intake (nuclear transformations per Bq), and
s	=	the source year.

The specific effective energy and the number of nuclear transformations are given by

$$SEE(T7S) = \frac{\sum_R y_R E_R w_R AF(T7S)}{M_t} \qquad \text{(Eq. 11.2)}$$

The summation is made over all the radiations (*R*) where

y_R	=	the yield of the radiation *R* per nuclear transformation (Bq^{-1} s^{-1});
E_R	=	the energy of the radiation *R* (MeV**)
w_R	=	the radiation weighting factor (Sv Gy^{-1}***);
AF (T7 S)	=	the absorbed fraction, that is, the fraction of the energy E_R emitted in the source organ *S* that is absorbed in the target organ *T* (unitless); and
M_T	=	the mass of the target organ (kg).

The number of nuclear transformations due to the activity $q_s(t)$ in the source organ is given by the following formula:

---

* one nuclear transformation = 1 Bq × 1 sec.

** 1 MeV = 1.602×10^{-13} Joules.

*** 1 Gy = 1 J kg^{-1} = 100 rad.

$$U_{nt,s} = \int_{t_o}^{t_o+t} q_s(t)dt \qquad \text{(Eq. 11.3)}$$

where

$q_s(t)$ = the activity as a function of time (obtained from the biokinetic model of each radionuclide), and

t = the period over which the doses will be delivered (50 years for adults and 70 years for children).

The biokinetic model is a combination of the gastrointestinal tract model and a metabolic model representing the bioaccumulation of each radionuclide in the body. In the gastrointestinal tract model, the transfer of material through the different segments of the tract is the same for all radionuclides. However, the absorption from the intestines to blood and the metabolism of a radionuclide in the body depend on the radionuclide and, in some cases, on its chemical form. A description of the gastrointestinal tract model and a presentation of the masses for the target organs are provided separately in Sections 11.1.1 and 11.1.3, since these components are not element-specific. On the other hand, the element-specific components such as the fraction absorbed from the gastrointestinal tract and the amount of radionuclide transferred to blood (denoted as f_1), are radionuclide-specific; metabolic models are discussed in Sections 11.2 to 11.6.

Three main sources of uncertainty can be identified in the ingestion dose for a unit intake: the number of nuclear transformations ($U_{nt,s}$) given by the biokinetic model; the fraction of energy emitted that is absorbed in the organ of interest [*AF(T7S),* discussed in Section 11.1.2]; and the mass of the target organ (M_T). The nuclear properties (half-lives, energy and yield of each radiation) for each radionuclide are considered accurate quantities, and no uncertainty is associated with them. Nuclear properties for all radionuclides of interest in this assessment are presented in Appendix 10A.

The dose to a specific organ is generally defined as the energy deposited by the radiation per unit mass of the organ. Various organ masses have large individual variation depending on gender, state of health, or just natural variability. A special technique was applied for treating the uncertainty associated with the variability of the mass of different organs. The technique was based on the observation that an individual can be simultaneously exposed to various radionuclides and exposure pathways; in other words, a given organ of the exposed individual will receive different amounts of energy from each radionuclide and exposure pathway. These amounts of energy are independent of each other, even though they are delivered to the same organ of a given mass. In a deterministic analysis, the organ doses are additive, but in a probabilistic uncertainty analysis they are not. To assess the uncertainty in the dose to a given organ, the total energy from different radionuclides and exposure pathways must be calculated first, and then the total is divided by the probability distribution describing the uncertainty in the organ mass. This method was

employed in the present study. Therefore, in addition to calculating the dose per unit intake (ingestion dose factor), the energy per unit intake (*EUI)* was estimated for each organ and for each radionuclide, using the following equation:

$$EUI_T = \sum_S U_{nt,S} \cdot \sum_R y_R E_R w_R AF(T7S) \qquad \text{(Eq. 11.4)}$$

The estimation of the EUI (Eq. 11.4) was performed in the framework of an uncertainty analysis. Probability distributions rather than point estimates have been produced for the energy per unit intake. The energy per unit intake was then used in the overall calculation to estimate the total energy deposited in a given organ for various radionuclides and exposure pathways.

Different approaches were employed to determine the dose factors and the energy per unit intake (Sections 11.2-11.6). The estimates of the ingestion dose factors are not used directly in the final dose calculations, but they are produced only for a comparison to the current ICRP values. To obtain the distributions of the energy per unit intake, the probability distributions for the ingestion dose factors are multiplied by point estimates of the organ masses for each organ of interest. Neither the dose factors nor the energies per unit intake reported in this section contain the uncertainty associated with the mass of the organ. Section 11.1.3 describes the treatment employed for the uncertainty in organ masses.

Finally, a sensitivity analysis was performed for each radionuclide to identify the contribution of various sources of uncertainty to the estimated energy per unit intake. Of the three sources of uncertainty discussed earlier, only the uncertainties in the biokinetic model are considered in this section. They are described as probability distributions for the number of nuclear transformations in a source organ. The uncertainties in the fraction of energy absorbed by the target organ were found to be small (Sections 11.1.2 and 11.2.6) as compared to the other uncertain components, and are therefore not included. The sensitivity analyses presented in the following sections show only the most important biokinetic parameters in internal dosimetry. As the uncertainty in the organ mass is not included, the sensitivity analysis does not contain the contribution of uncertainty caused by the organ masses. The importance of the organ mass to the uncertainty in the dose to a target organ is obtained from the sensitivity analysis on the total dose for that organ.

11.1.1 The Gastrointestinal Tract

After the radionuclides mixed in foods are ingested, they pass through stomach, then through the intestines, until they are finally excreted. The ICRP (1979) gastrointestinal (GI) tract model (Figure 11.1.1) consists of four compartments: the stomach, the small intestine, the upper large intestine, and the lower large intestine. These compartments define also the main organs for which the radiation doses are calculated. The doses to the GI tract organs are produced by the radionuclides

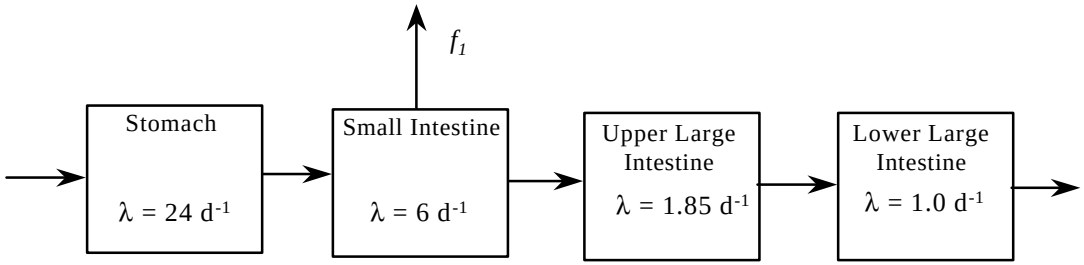


Figure 11.1.1 Gastrointestinal Tract Model used for all radionuclides (ICRP, 1979).

in the contents of the GI tract, by radionuclides accumulated in the walls of these organs, and by irradiation from radionuclides in other organs.

The residence time of food within each section of the GI tract differs from one type of food to another, and for the same type of food from one individual to another. Solid foods (e.g., bread, cereals, meats) have a longer residence time than softer foods (e.g., yogurt, ice-cream, puddings) or liquid foods (e.g., soups or milk). A review of the GI tract kinetics for the purpose of radiation dosimetry was performed by Eve (1966) and Dolphin and Eve (1966). These reviews are the basis for the ICRP (1979) GI tract model.

During passage through the gastrointestinal tract, a radionuclide is partially or completely absorbed by the blood. The fraction of the ingested amount that is transferred into blood is referred to as "the absorption fraction" (f_1). This parameter should not be confused with the fraction of energy emitted in the source organ that is absorbed in a target organ, which is called the "absorbed fraction" ($AF(T \leftarrow S)$). The absorbed fraction was introduced in Section 11.1 and is discussed in Section 11.1.2.

For most radionuclides, the ICRP model assumes that absorption occurs only in the small intestine. The absorption fraction (f_1) depends on the chemical form of each radionuclide. Hence, this parameter is radionuclide-specific, and it is discussed in the sections describing each isotope.

The ICRP developed the gastrointestinal tract model for the derivation of the intake limits for radiation workers; the model was not intended to address the natural variability between individuals in a population. The transfer rates (l) in the ICRP model are defined as the inverse of the mean residence times of the material in each compartment. The mean residence times used by ICRP (1979) for each compartment of the gastrointestinal tract model are as follows: stomach, 1 hr; small intestine, 4 hrs.; upper large intestine, 13 hrs.; and lower large intestine, 24 hrs. These values are based on the review of the experimental data reported by Eve (1966). The same paper reports that the observed residence times vary from 0.4 to 2.0 hours for stomach, from 1 to 7 hours for the small intestine, from 6 to 22 hours for the upper large intestine, and from 15 to 72 hours for the lower large intestine. These observed ranges are the basis for the definition of the probability distributions describing the uncertainty in the residence times of foods in the gastrointestinal tract.

The uncertainty in the transfer rates appears to be within a factor of about two[3]. Table 11.1.1 presents the transfer rates and the uncertainty ranges selected for each transfer rate. Triangular probability distribution functions were subjectively assigned to express the uncertainty in the transfer rates.

[3] "A factor of *n*" (in this case two) indicates a number *n* that defines an uncertainty range about a quantity *x* having a minimum of *x* divided by *n* and a maximum of *x* times *n*.

11.1.2 Fraction of Energy Emitted from the Source that is Absorbed by the Target Organ

A second source of uncertainty is the fraction of energy emitted in a given organ, *S*, that is absorbed in another organ of interest. This quantity is called the "absorbed fraction" and is denoted as follows:

$$AF(T7 \leftarrow S)$$

Electrons do not usually have enough energy to travel from one organ to another, and most of the electron emissions are totally absorbed within the organ in which they are emitted. Therefore, the uncertainty associated with the energy deposited in a given organ by electron emission (â-radiation) can be considered as negligible with respect to other more dominant sources of uncertainty.

Table 11.1.1 Uncertainties associated with the transfer rates (λ's in Figure 11.1.1) between various components of the gastrointestinal tract

From	To	Minimum Value ($days^{-1}$)	Mode ($days^{-1}$)	Maximum Value ($days^{-1}$)	Distribution
Stomach	Small Intestine	12.0	24.0	57.6	Triangular
Small Intestine	Upper Large Intestine	3.4	6.0	24.0	Triangular
Small Intestine	Plasma	3.4	6.0	24.0	Triangular
Upper Large Intestine	Lower Large Intestine	1.1	1.85	4.0	Triangular
Lower Large Intestine	Feces	0.3	1.0	1.6	Triangular

The absorbed fraction for photons (gamma radiation) is usually obtained by very complex Monte Carlo simulations of the radiation transport from one organ to another. The human body is represented as a mathematical ph`antom in which organs are described by their mass, idealized shape, position, and composition. The absorbed fractions are calculated generally to an error level below 20%, but not larger than 50% (Stansbury, 1994; Snyder et al., 1969). An investigation of the effect of the errors in the

absorbed fraction was performed for ^{137}Cs (see Section 11.2.6). The uncertainty in the ingestion dose was found to be relatively insensitive to the error in the absorbed fraction. For example, errors of 25%, 50% and a factor of 2 in the absorbed fraction (*AF*) were found to have contributed 0.2%, 1.8% and 8.1% respectively to the total uncertainty in the ingestion dose from ^{137}Cs. These errors are generally lower than most of the other important sources of uncertainty, such as the parameters of the biokinetic model (about 30%) or the absorption from the gastrointestinal tract to the blood (about 2.5%; see Section 11.2.6). Based on this finding, no uncertainty was assigned to the absorbed fraction for all radionuclides that distribute fairly uniformly in the body (e.g., ^{137}Cs).

Radionuclides such as ^{60}Co and ^{106}Ru accumulate in certain organs shortly after ingestion, and they become more uniformly distributed in the body at later times. The absorbed fraction was also assumed to be accurately known for these radionuclides. Strontium-90 is a special case because it emits only nonpenetrating (beta) radiation that is locally absorbed; no uncertainty was associated with the absorbed fraction for ^{90}Sr. Iodine-131 accumulates in the thyroid, which is a relatively small organ. Therefore, part of the penetrating radiation emitted by ^{131}I will escape the thyroid without any energy deposition. The amount of energy that escapes depends on the size, and therefore the mass of the thyroid. Since the thyroid is one of the organs with very large interindividual variation, the absorbed fraction will be affected by some uncertainty. The uncertainty in the energy absorbed fraction for iodine was estimated as a function of the uncertainty in the mass of the thyroid, as described in Section 11.6.

11.1.3 Organ Masses

The dose to a specific organ is generally defined as the energy deposited by the radiation divided by the mass of the specific organ. The masses of various organs vary widely from one individual to another according to age, gender, state of health, or just natural variability. This section addresses the magnitude of the uncertainties in the organ masses.

The uncertainty in the organ masses is introduced by interindividual variability, compounded by errors in organ measurement techniques (e.g., autopsy or ultrasonography). The uncertainties and the associated distributions for the organ masses, obtained from the literature, are presented in Table 11.1.2.

Table 11.1.2 Masses of the target organs (kg) used in determining the dose.

Organ	ICRP Value	Parameter 1	Parameter 2	Shape	Parameter Information
Adrenals	1.4×10^{-2}	1.4×10^{-2}	2.8×10^{-3}	Lognormal	Mean and standard deviation [a]
Bladder	4.5×10^{-2}	3.0×10^{-2}	6.0×10^{-2}	Uniform	Minimum and Maximum values [a]
Bone Surface	1.2×10^{-1}	6.0×10^{-2}	2.4×10^{-1}	Loguniform	Minimum and Maximum (a factor of 2) [b]
Brain	1.4×10^{0}	1.4×10^{0}	2.4×10^{-2}	Lognormal	Mean and standard deviation [a]
Breast	3.6×10^{-1}	3.6×10^{-1}	2.8×10^{-1}	Normal	Mean and 2.5%-tile [a]
Digestive System					
Oral Cavity	1.0×10^{-1}	NA*	NA*	NA*	NA*
Esophagus	3.7×10^{-2}	NA*	NA*	NA*	NA*
Gallbladder	1.0×10^{-2}	NA*	NA*	NA*	NA*
Stomach	1.5×10^{-1}	1.2×10^{-1}	1.6×10^{-1}	Normal	10%-tile and 90%-tile [a]
Small Intestine	6.4×10^{-1}	5.4×10^{-1}	8.4×10^{-1}	Normal	10%-tile and 90%-tile
Upper Large Intestine	2.1×10^{-1}	1.7×10^{-1}	2.4×10^{-1}	Normal	10%-tile and 90%-tile
Colon	1.6×10^{-1}	1.2×10^{-1}	2.0×10^{-1}	Normal	10%-tile and 90%-tile
Rectum	3.5×10^{-2}	NA*	NA*	NA*	NA*
Kidneys	3.1×10^{-1}	2.4×10^{-1}	3.8×10^{-1}	Uniform	Minimum and maximum [a]
Liver	1.8×10^{0}	1.4×10^{0}	2.3×10^{0}	Lognormal	10%-tile and 90%-tile [a]
Lungs	1.0×10^{0}	1.1×10^{0}	1.4×10^{-1}	Lognormal	Mean and standard deviation [c]
Muscle	2.8×10^{1}	2.0×10^{-2}	3.6×10^{-2}	Uniform	Minimum and maximum [a]
Ovaries	1.1×10^{-2}	1.1×10^{-2}	1.3×10^{-3}	Lognormal	Mean and standard deviation [a]
Pancreas	1.0×10^{-1}	1.0×10^{-1}	1.9×10^{-2}	Lognormal	Mean and standard deviation [a]
Prostate	1.6×10^{-2}	NA*	NA*	NA*	NA*
Red Bone Marrow	1.5×10^{0}	1.6×10^{0}	3.4×10^{-1}	Normal	Mean and standard deviation [d]
Skin	2.6×10^{0}	NA**	NA**	NA**	NA**
Spleen	1.8×10^{-1}	NA*	NA*	NA*	NA*
Testes	3.5×10^{-2}	1.3×10^{-2}	5.4×10^{-2}	Normal	2.5%-tile and 97.5%-tile [a]
Thyroid	2.0×10^{-1}	1.5×10^{-1}	1.5×10^{0}	Lognormal	Median and geometric standard deviation [e]
Thymus	2.0×10^{-2}	2.0×10^{-2}	8.3×10^{-3}	Lognormal	Mean and standard deviation [a]
Uterus	8.0×10^{-2}	NA**	NA**	NA**	NA**
Remainder	†	†	†	†	NA
Whole Body	6.9×10^{1}	6.9×10^{1}	1.1×10^{1}	Lognormal	Mean and standard deviation [a]

[a] ICRP (1992). [b] Leggett (1997a). [c] Spitzka (1904). [d] Ellis (1961). Woodard and Holodny (1960). [e] Killough and Eckerman (1986) and Section 11.6.

† The "Remainder" is calculated as the mass of the whole body minus the sum of the masses of the target organs.

* No uncertainties were developed for these organs. Oral cavity, esophagus, gallbladder, rectum and prostate were added to the ICRP list of organs for the purpose of evaluating the total risk of cancer incidence (Section 12.2.6).

** No risk was estimated for uterus, because the uterus has a negative dose-response relationship. No risk was calculated for skin because of the lack of a dose-response for melanoma.

The masses of the target organs used in this study are obtained primarily from the ICRP report on Reference Man (ICRP, 1975). This comprehensive report summarizes the data available from various measurements in adults and provides a mean value and a range for different organ masses. In the present study, to express the uncertainty in the organ masses, probability distribution functions were defined using the mean as a central value and the range as the limits of the distribution. For kidneys and testes, the volume of the organ instead of the mass was provided, and the mass was estimated using the density of each particular tissue.

For definition of the probability distribution functions (Table 11.1.2.), the following general rules were applied according to the data provided in various literature sources as cited in Table 11.1.2.

(a) If a mean and a symmetrical range about the mean were provided, a uniform distribution between the limits of the range was assigned.

(b) If a mean and a standard deviation were provided, a lognormal distribution was assigned.

(c) If a mean and a tight range or a small standard deviation were provided, a normal distribution was assigned.

(d) If an uncertainty factor about a central value was provided, a loguniform distribution was assigned with the limits obtained by applying the uncertainty factor.

For bone surface, red bone marrow, lungs and intestines, no range was provided in ICRP (1992). For these organs, data published in scientific papers were collected and analyzed. For bone surface, in particular, no data were available, since this type of “organ” is used mainly in radiation dosimetry. In this case, the expert opinion of Dr. Leggett (1997a, b), ICRP member, was elicited. For red bone marrow, studies by Ellis (1961) and Woodard and Holodny (1960) indicated that the mass of the bone marrow is about 4.6% (3.4% - 5.9%) of the total body weight. The red bone marrow accounts for about half of the bone marrow mass, while the rest is considered to be yellow bone marrow. The mass of the red bone marrow was calculated using the distribution for the body weight (Table 11.1.2) and a uniform distribution for the fraction of red bone marrow presented above. The resulting Monte Carlo samples for the red bone marrow were fitted to a normal distribution having a mean of 1.59 kg and a standard deviation of 0.34 kg. The mass of the lungs was determined using data from Spitzka (1904). Individual masses were obtained individually for the right and left lung. Using Spitzka's (1904) data, the mass of both lungs was determined to be 1.05 kg with a standard deviation of 0.14 kg. The mass of the intestines was determined using medical examiner's studies conducted in New York. The studies included 61 individuals (49 males and 12 females). Using the reported mean and standard deviation for all individuals, the 10th and 90th percentiles were calculated for the small and large intestines (Table 11.1.2; Tipton and Cook, 1969).

ICRP lists no mass for the oral cavity. However, ICRP lists the mass of the tongue as 70 g, and the mouth lining and pharynx as 30 g, thus giving a mass of 100 g for the oral cavity. Also, ICRP does not cite a value for the mass of the rectum. A value of 70 g is given for the rectum plus the sigmoid colon. The rectum makes up approximately half of this mass. Therefore, a value of 35 g is taken as the mass of the rectum.

Differences between genders are observed in terms of the gender-specific organs. Adult male organ masses were used in the determination of the doses, with the exception of the uterus, ovaries, and breasts. The doses to children were examined only in the case of the thyroid being irradiated by ^{131}I, because children are the critical population group for this exposure scenario. The variability in the mass of thyroid is presented in Sect. 11.6, together with information regarding doses due to radioactive iodine.

11.2 Cesium-137

Cesium-137 was determined during the initial screening analysis to be an important radioactive isotope released from the Oak Ridge Reservation via White Oak Creek. As a result, the dose contribution of ^{137}Cs to the overall dose possibly received by adults living along the Clinch River was required. The dose factor is one element of uncertainty in estimating dose. The purpose of this section is to assess the uncertainty in the DF for ^{137}Cs, to determine the most important factors contributing to the uncertainty in the DF for ^{137}Cs (sensitivity analysis), and to describe the way in which this uncertainty will be treated in the risk assessment.

11.2.1 Background

Cesium-137 is a thoroughly studied fission product. The behavior of ^{137}Cs in the human body is particularly well understood, and the ingestion doses per unit of activity ingested are fairly well known. Appendix 10-A provides a description of the main nuclear properties of ^{137}Cs.

11.2.2 Absorbed Fraction

The energy emitted by each nuclear transformation is completely absorbed in the body if beta radiation occurs. On the other hand, only about 34% of the emitted energy is absorbed in the whole body if a 662 KeV gamma ray occurs (Schwartz and Dunning, 1982). The amount of emitted gamma energy absorbed in a given organ varies from 1% for small organs to 17% for the liver and 28% for the muscles.

The absorbed fraction (*AF*) for gamma emissions is usually obtained by Monte Carlo methods, which simulate the transport of radiation from a source organ to a target organ. However, this type of Monte Carlo analysis requires a major computational effort. Therefore, an idealized mathematical phantom of the human body containing organ masses, organ positions, and idealized organ shapes is used, and all combinations of source and target organs are taken into account. The absorbed fraction (*AF*) is generally calculated at an error level below 20%, but not larger than 50% (Stansbury, 1994; Snyder et al., 1969). The uncertainty in the absorbed fraction is difficult to assess because of the complexity of the human body,

even when the body is presented as an idealized phantom. The effects of different levels of uncertainty in the final results are discussed in Section 11.2.5. In this section, it is shown that uncertainty in the *AF* can be neglected

11.2.3 Biokinetics of ^{137}Cs in the Human Body

Cesium biokinetics is correlated with the potassium distribution and movement in the human body, because cesium and potassium have close chemical similarities. All cells in the human body contain traceable amounts of potassium in their cytoplasm. Potassium is assumed to represent 0.2% of the total body weight of Reference Man (70 kg), which is equivalent to 140 grams (ICRP, 1975).

11.2.3.1 Absorption from the Gastrointestinal Tract to Blood

Most studies (ICRP, 1975; 1989; Leggett, 1986) agree that cesium in the inorganic form is completely (100%) and rapidly absorbed by the blood from the gastrointestinal tract. However, for ^{137}Cs contained in meat ingested by humans, the fraction of ^{137}Cs transferred to the blood was found to be marginally lower, varying from 72% to 85% with a mean of 80.4% (Talbot et al., 1993). In the present study, values from 80% to 100% were considered plausible for the percentage of ^{137}Cs in all ingested material absorbed from the gastrointestinal tract. A triangular distribution was assigned for this parameter with a minimum value of 80%, a mode of 100%, and a maximum of 100% (Table 11.2.1). The selected values were based on those reported in the literature (ICRP, 1975; 1989; Leggett, 1986; Talbot et al., 1993).

Since most of the absorption occurs rapidly in the stomach and the small intestine, practically no cesium passes through the large intestine, and thus the irradiation of the gastrointestinal tract from cesium in the gastrointestinal tract is minimal.

11.2.3.2 Retention and Elimination of ^{137}Cs

The blood transports cesium to all organs and tissues, where it may be absorbed or excreted. Cesium is nearly uniformly distributed throughout the body and is excreted in both urine and feces. The amount excreted in urine is usually 4 times larger than the amount excreted in feces.

Table 11.2.1 Parameters of the biokinetic model for ^{137}Cs.

Absorption from the Gut f_1	**Units** **%**	**Minimum** **80%**	**Mode** **100%**	**Maximum** **100%**	**Defined Distribution** **Triangular**	**ICRP Default Value[a]** **100%**
Biokinetic Parameters Used in This Study	Units	Mean	Standard Deviation		Fit Distribution	ICRP Default Value[a]
a		0.11	0.04		Lognormal	0.1
T_1	days	1.9	0.7		Lognormal	2
T_2	days	101.8	15.8		Normal	110
Biokinetic Parameters Used in Other Studies[b]	Units	Mean	Standard Deviation		Fit Distribution	ICRP Default Value[a]
a		0.12	0.048		Lognormal or normal[c]	0.1
T_1	days	1.15	0.37		Lognormal or normal[c]	2
T_2	days	100.0	27.0		Lognormal or normal	110

[a] ICRP, 1979; 1989.
[b] Schwartz and Dunning, 1982.
[c] Both lognormal and normal have been used by Schwartz and Dunning (1982).

Since the half-life of ^{137}Cs's decay product ^{137m}Ba is very short (2.55 minutes), a reasonable assumption is that the sites of decay for ^{137m}Ba should be the same as those of its parent. The issue of sites of decay is important because the majority of the energy responsible for the delivered dose comes from ^{137m}Ba and not from ^{137}Cs. However, experimental evidence suggests that ^{137m}Ba migrates from its parent (Wasserman et al., 1959). By determining the distribution of ^{137}Cs and ^{137m}Ba in rats within a few minutes after death, Wasserman et al. (1959) demonstrated some dissociation of ^{137m}Ba from administered ^{137}Cs. Barium-137m should follow the same biokinetic paths as ^{137}Ba, which has some affinity for bone and is also rapidly transferred from blood to the intestinal content. This evidence, however, has not yet been incorporated into the ICRP standard models. The doses to the bone and to the gastrointestinal tract organs might change as a result of the change of the standard models. Such a modification of the models was not considered in this work.

The currently accepted biokinetic model (Figure 11.2.1) consists of two compartments for the total body (ICRP 1979; 1989). The fraction transferred to the fast compartment is denoted as ***a***; the default value recommended by ICRP (1979; 1989) is 10%. The default values (ICRP, 1979; 1989) for the biological half-lives for the fast and the slow compartments, T_1 and T_2, are 2 days and 110 days, respectively. The longest biological half-life in any of these compartments is much smaller than the radioactive half-life of ^{137}Cs (30 years); in most internal dosimetry calculations for ^{137}Cs, the radioactive decay can be neglected.

Parameters a, T_1, and T_2 are dependent on each other (Table 11.2.1 provides the numerical values for parameters a, T_1, and T_2). This dependence must be taken into consideration when working in an uncertainty analysis framework. Leggett (1986), using experimental data, derived the relationship between the above parameters and the amount of potassium in the total body (K_t).

The relationships for the adult male are as follows:

$$a = 0.81 \cdot e^{-0.014 \cdot K_t} \qquad R = 0.92 \qquad \text{(Eq. 11.2.1a)}$$

$$T_1 = 18.0 \cdot e^{-0.016 \cdot K_t} \qquad R = 0.87 \qquad \text{(Eq. 11.2.1b)}$$

$$T_2 = -1.22 + 0.72 \cdot K_t \qquad R = 0.91 \qquad \text{(Eq.11.2.1c)}$$

where R = coefficient of determination from the statistical test.

The amount of potassium in the total body, K_t (grams), was determined using the concentration of potassium in the human body (g kg^{-1} body weight) and the human body mass (kg) provided by ICRP (1975; Table 11.2.2). The uncertainty in the total amount of potassium (K_t) is calculated by error propagation.

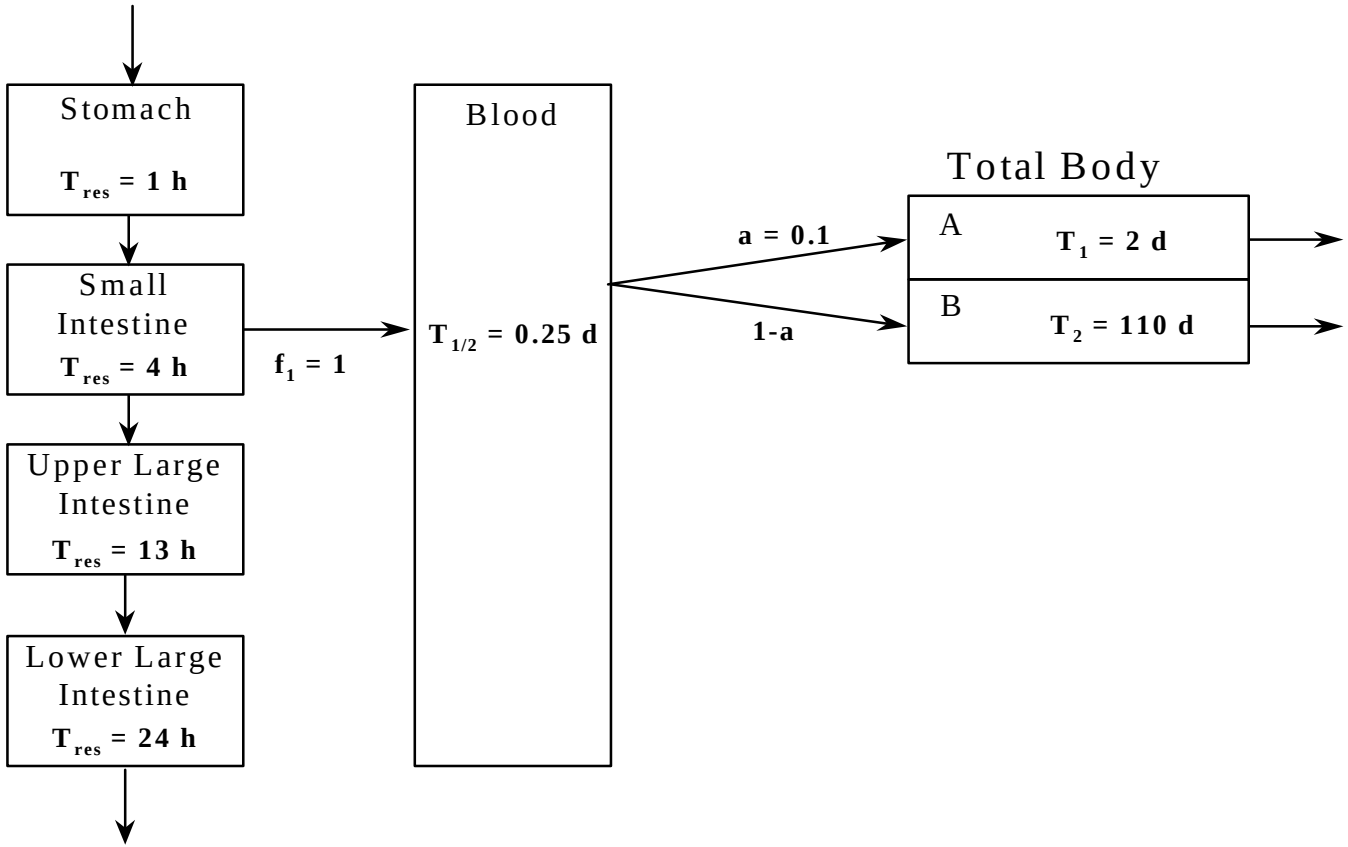


Figure 11.2.1 Metabolic model for ^{137}Cesium (adult values provided) (ICRP, 1977).

Table 11.2.2 The body mass and the concentration of potassium in Reference Man (ICRP, 1977) used to calculate the total amount of potassium in the human body.

Parameter	Units	Mean	Standard Deviation	Distribution
Body mass	kg	68.8	10.5	Normal
Concentration of potassium in the human body	$g\,kg^{-1}_{body\,weight}$	2.1		Constant

One of the most complete studies related to the uncertainty of the model parameters was performed by Schwartz and Dunning (1982). The distributions reported by Schwartz and Dunning (1982) for parameters a, T_1, and T_2 are given in Table 11.2.1.

11.2.4 Results and Conclusions

The uncertainty in the dose factors for ^{137}Cs was expressed as a multiplicative uncertainty factor that applies to all organs. As a result of the calculations, median values of the dose factor were found to be about a factor of 14.5% lower than the default ICRP values (Table 11.2.3; Figure 11.2.2). About a six percent difference is due to consideration of a slightly lower transfer of ^{137}Cs from the gut to the blood (Table 11.2.1). The other eight percent difference comes from the half-life in the long-term ^{137}Cs storage compartment, which was found to be lower (102 days) than the ICRP default value (110 days). The overall uncertainty introduced is a factor of 1.4 (calculated as the ratio between the 97.5%-tile and 50%-tile of the output distribution), which is consistent with previously estimated values (Schwartz and Dunning, 1982; Bouville et al., 1994). Using the same approach, with the uncertainties suggested by Schwartz and Dunning (1982) for a, T_1, T_2 and no correlation between them, an uncertainty factor of 1.7 is obtained. This uncertainty can be considered small compared to other components influencing the final dose and risk estimates (such as the bioconcentration factor for the accumulation of ^{137}Cs in fish or the intake of ^{137}Cs by humans).

The results are also presented in terms of the energy deposited in various organs after the ingestion of 1 Bq of radioactive material (Table 11.2.4). These energies were used to estimate the total dose of ^{137}Cs for each organ.

Table 11.2.3 Organ-specific dose conversion factors for adults from ingestion of ^{137}Cs. The ICRP factors are compared to the distributions derived in this work.

Target Organ	**Present Study (Sv Bq^{-1})**			**ICRP Reference Man[a] (Sv Bq^{-1})**
	95% Subjective Confidence Interval			
	lower bound	**central value**	**upper bound**	
Adrenals	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Bladder	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Bone surface	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Brain	6.5×10^{-9}	1.0×10^{-8}	1.4×10^{-8}	**1.2×10^{-8}**
Breast	6.0×10^{-9}	9.4×10^{-9}	1.3×10^{-8}	**1.1×10^{-8}**
Digestive System				
Oral Cavity	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Esophagus	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Stomach	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Small Intestine	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Upper Large Intestine	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Colon	9.3×10^{-9}	1.5×10^{-8}	2.0×10^{-8}	**1.7×10^{-8}**
Rectum	9.3×10^{-9}	1.5×10^{-8}	2.0×10^{-8}	**1.7×10^{-8}**
Gallbladder	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Kidneys	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Liver	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Lungs	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Muscle	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Ovaries	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Pancreas	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Prostate	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Red Bone Marrow	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Skin	6.0×10^{-9}	9.4×10^{-9}	1.3×10^{-8}	**1.1×10^{-8}**
Spleen	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Testes	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Thymus	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Thyroid	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Uterus	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**
Remainder[b]	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}	**1.3×10^{-8}**
Effective (ICRP)[c]	7.6×10^{-9}	1.2×10^{-8}	1.7×10^{-8}	**1.4×10^{-8}**

[a]ICRP Publication 67 (ICRP,1993).

[b]Based on the value for the remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

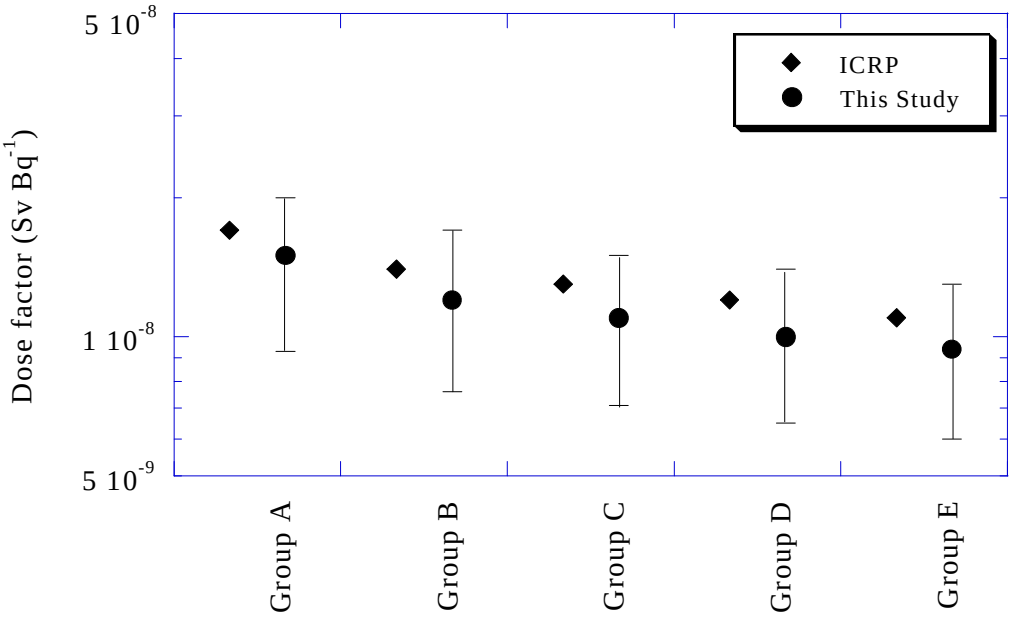
[c]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0.

Table 11.2.4 Energy deposited in various organs from ingestion of 1 Bq of ^{137}Cs by adults.

Target Organ	Energy Deposited in Each Target from a Unit Intake [J/Bq]		
	95% subjective confidence interval		
	lower bound	central value	upper bound
Adrenals	1.1×10^{-10}	1.7×10^{-10}	2.3×10^{-10}
Bladder	3.4×10^{-10}	5.4×10^{-10}	7.5×10^{-10}
Bone surface	9.2×10^{-10}	1.4×10^{-9}	2.0×10^{-9}
Brain	9.2×10^{-9}	1.4×10^{-8}	2.0×10^{-8}
Breast	2.2×10^{-9}	3.4×10^{-9}	4.7×10^{-9}
Digestive System			
Oral Cavity	7.1×10^{-10}	1.1×10^{-9}	1.5×10^{-9}
Esophagus	2.6×10^{-10}	4.1×10^{-10}	5.7×10^{-10}
Stomach	1.1×10^{-9}	1.7×10^{-9}	2.3×10^{-9}
Small intestine	4.9×10^{-9}	7.7×10^{-9}	1.1×10^{-8}
Upper Large Intestine	1.6×10^{-9}	2.5×10^{-9}	3.5×10^{-9}
Lower Large Intestine	1.5×10^{-9}	2.3×10^{-9}	3.2×10^{-9}
Rectum	3.2×10^{-10}	5.1×10^{-10}	7.1×10^{-10}
Gallbladder	7.1×10^{-11}	1.1×10^{-10}	1.5×10^{-10}
Kidneys	2.4×10^{-9}	3.7×10^{-9}	5.2×10^{-9}
Liver	1.4×10^{-8}	2.2×10^{-8}	3.0×10^{-8}
Lungs	7.1×10^{-9}	1.1×10^{-8}	1.5×10^{-8}
Muscle	2.0×10^{-7}	3.1×10^{-7}	4.3×10^{-7}
Ovaries	8.4×10^{-11}	1.3×10^{-10}	1.8×10^{-10}
Pancreas	7.6×10^{-10}	1.2×10^{-9}	1.7×10^{-9}
Prostate	1.1×10^{-10}	1.8×10^{-10}	2.5×10^{-10}
Red Bone Marrow	1.1×10^{-8}	1.7×10^{-8}	2.3×10^{-8}
Skin	1.6×10^{-8}	2.4×10^{-8}	3.4×10^{-8}
Spleen	1.4×10^{-9}	2.2×10^{-9}	3.0×10^{-9}
Testes	2.5×10^{-10}	3.9×10^{-10}	5.4×10^{-10}
Thymus	1.4×10^{-10}	2.2×10^{-10}	3.1×10^{-10}
Thyroid	1.5×10^{-10}	2.4×10^{-10}	3.3×10^{-10}
Uterus	6.1×10^{-10}	9.6×10^{-10}	1.3×10^{-9}
Remainder[a]	4.1×10^{-7}	6.5×10^{-7}	9.1×10^{-7}
Effective (ICRP)[b]	5.3×10^{-7}	8.3×10^{-7}	1.1×10^{-6}

[a]Based on the value for remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

[b]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0.



Group A = Colon, Rectum

Group B = Adrenals, Bladder, Bone Surface, Small Intestine, Upper Large Intestine, Kidneys, Liver, Ovaries, Pancreas, Spleen, Thyroid, Uterus

Group C = Oral Cavity, Esophagus, Stomach, Lungs, Muscle,Prostate, Red Bone Marrow,Testes, Thymus

Group D = Brain

Group E = Breast, Skin

Figure 11.2.2 Comparison of the ingestion dose factors for ^{137}Cs calculated in this study to the ICRP (1993) values.

11.2.5 Sensitivity Analysis

The contribution of the biokinetic model to the uncertainty in the dose factor was separated into different components (Figure 11.2.3). The absorption from the gut to the blood has a minimal influence, while the biokinetic parameters are the dominant source of uncertainty.

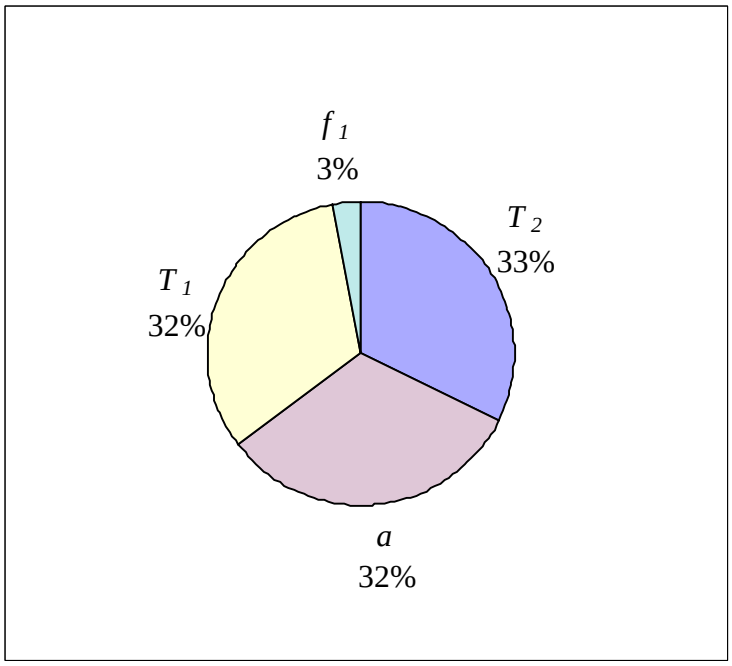
If the biokinetic parameters are independent, a small variation in the long-term half-life (T_2) or in the fraction *a* would produce a larger variation in the dose factors than a similar small variation in the short-term half-life (T_1). However, the three parameters are related by the total amount of potassium in the body (Eq. 11.2.1). Because this relationship was considered, the sensitivity analysis showed that each parameter has practically the same contribution to the uncertainty in the dose factors (Figure 11.2.2).

To investigate the contribution of uncertainty in the absorbed fraction (*AF*, Section 11.2.3) to the total uncertainty in the dose factors, a test was run for the whole-body dose factor, keeping the biokinetic parameters, but varying the whole-body absorbed fraction (*AF*). Arbitrary error levels of a factor of 1.25, 1.5 and 2.0 were selected for the absorbed fraction (*AF*), and loguniform distributions were assigned. The results (Table 11.2.5) indicated that even at an error level of 2.0, the contribution of the uncertainty in the absorbed fraction (*AF*) is less than 10% of the total uncertainty in the dose factor. Since the absorbed fraction (*AF*) is generally calculated to an error level below 20%, but not larger than 50% (Stansbury, 1994; Snyder et al., 1969), the uncertainty in the absorbed fraction (*AF*) can be neglected. Due to the similarities in the mechanism of energy deposition for ^{137}Cs, the same results can be extended to all other organs. Moreover, the same results should be valid for other radionuclides that distribute fairly uniformly in most of the organs of the body (^{60}Co, ^{106}Ru), even though they emit more than one gamma ray.

Table 11.2.5 Contribution of uncertainty in the absorbed fraction (*AF*) to the uncertainty in the whole-body dose factor for ingestion of ^{137}Cs.

Sensitivity Data	**Uncertainty in the Absorbed Fraction**		
	Factor of 1.25	**Factor of 1.5**	**Factor of 2**
T_1	32.5%	32.0%	29.8%
T_2	32.5%	32.0%	29.8%
a	32.5%	32.0%	29.8%
f_1	2.4%	2.3%	2.5%
AF	0.2%	1.8%	8.1%

LEGEND:
a = fraction transferred to the fast compartment
T_1 = biological half-life for the fast compartment
T_2 = biological half-life for the slow compartment
f_1 = fraction absorbed from the gastrointestinal tract to blood



LEGEND:

a	fraction transfered to the fast compartment
T_1	biological half-life for the fast compartment
T_2	biological half-life for the slow compartment
f_1	absorption fraction for the gastrointestinal tract to blood

Figure 11.2.3 Contribution to the total uncertainty of different biokinetic parameters for ^{137}Cs.

11.3 Strontium-90

Strontium-90 was determined during the initial screening analysis to be an important radioactive element released from the Oak Ridge Reservation via White Oak Creek. Therefore, the contribution of ^{90}Sr to the overall dose possibly received by target individuals living along the Clinch River was required. The dose factor (DF) is one element of uncertainty in the dose. The purpose of this section is to assess the uncertainty in the DF, to determine the most important factors contributing to the uncertainty in the DF for ^{90}Sr (sensitivity analysis), and to describe the way in which this uncertainty will be treated in the risk assessment.

11.3.1 Background Information

Strontium (Sr) is the thirty-eighth element on the periodic table and was originally found in Scotland in the village of Strontium where the mineral form, strontianite, was abundant (Wyllie in Lenihan et al., 1967). This element has 18 isotopes, which include ^{85}Sr, ^{89}Sr, and ^{90}Sr (Weast, 1968). Strontium belongs to a family of elements known as the alkaline earths, and its properties are similar to those of calcium, barium, and radium (Coughtrey and Thorne, 1983). Appendix 10-A provides information on the main nuclear properties of ^{90}Sr.

11.3.2 Absorbed Fraction

The fraction of the energy emitted by a source organ and absorbed in a target organ is known as the absorbed fraction (*AF*). Strontium and calcium behave similarly in the body and are absorbed in a similar manner; data on the distribution of strontium in the human body indicate that the element has a high affinity for mineral bone but is not significantly concentrated in any other organ or tissue of the body (Coughtrey and Thorne, 1983). For strontium dosimetry, therefore, the "source regions" are trabecular bone and the cortical bone, and the primary "target regions" are the bone surface and the red bone marrow (Table 11.3.1). A description of these tissues is presented below (Section 11.3.4).

For tissues other than bone, the energy of the electrons is considered to be absorbed entirely in the target cells, because the range of these "low energy" electrons is less than 4.3 mm (Turner, 1992). Bone tissue is heterogeneous; that is, some bone structures are so thin that electrons can penetrate through them and escape into the surrounding tissues. Thus, only part of the energy carried by the electrons is absorbed in the target tissue. ICRP Publication 30 (ICRP, 1979) recommends the following absorbed fractions for the skeleton for different combinations of source and target tissues. These fractions were used in our estimation of the dose factors for ^{90}Sr.

Table 11.3.1 Absorbed fractions for the skeleton based on bone type, emitter[a] type, and emitter energy (Eckerman, 1994).

Source Region	Target Tissue	Volume Emitter[a]	Surface Emitter[a]	
			$\overline{E_\beta} \# 0.2\ MeV$	$\overline{E_\beta} > 0.2\ MeV$
Trabecular Bone	Bone Surface	0.025	0.25	0.025
Trabecular Bone	Red Bone Marrow	0.35	0.5	0.5
Cortical Bone	Bone Surface	0.015	0.25	0.015
Cortical Bone	Red Bone Marrow	0.0[b]	0.0[b]	0.0[b]

[a] Strontium accumulates both in the volume and on the surface of the source regions.
[b] The red bone marrow is totally associated with trabecular bone and is beyond the range of beta particles emitted within cortical bone.

11.3.3 Biokinetics of ^{90}Sr in the Human Body

Bone and its components are the critical organs for strontium dosimetry. Data demonstrate the high affinity of mineral bone for strontium, but they also show that the element is not significantly concentrated in any other organ or tissue (Coughtrey and Thorne, 1983). Bone is divided into two compartments: cortical and trabecular, which are further divided into bone surface and bone volume. Cortical bone or compact bone is the hard crystalline bone that provides the structure (ICRP, 1975). This type of bone is found in the shafts of the long bones. Trabecular or cancellous (spongy) bone contains the marrow (ICRP, 1975). The marrow is held in place by splinters of bone mineral. The spongy bone is found primarily in the ends of the long bones. Cortical bone comprises about 80% of the adult skeleton and trabecular bone about 20%, by volume, by mass, and by calcium content (ICRP, 1975).

When strontium is ingested, a substantial fraction of the ingested activity makes its way from the gastrointestinal tract into the plasma (see Figure 11.3.1). Once in the plasma, the activity is distributed primarily to the bones. Mobilization of bone mineral is accomplished with both short-term and long-term mechanisms of accretion, loss, and exchange. Rapidly exchangeable activity in bone is associated with the bone surface, which is treated as a uniformly mixed compartment that exchanges activity with the blood (Leggett, 1992).

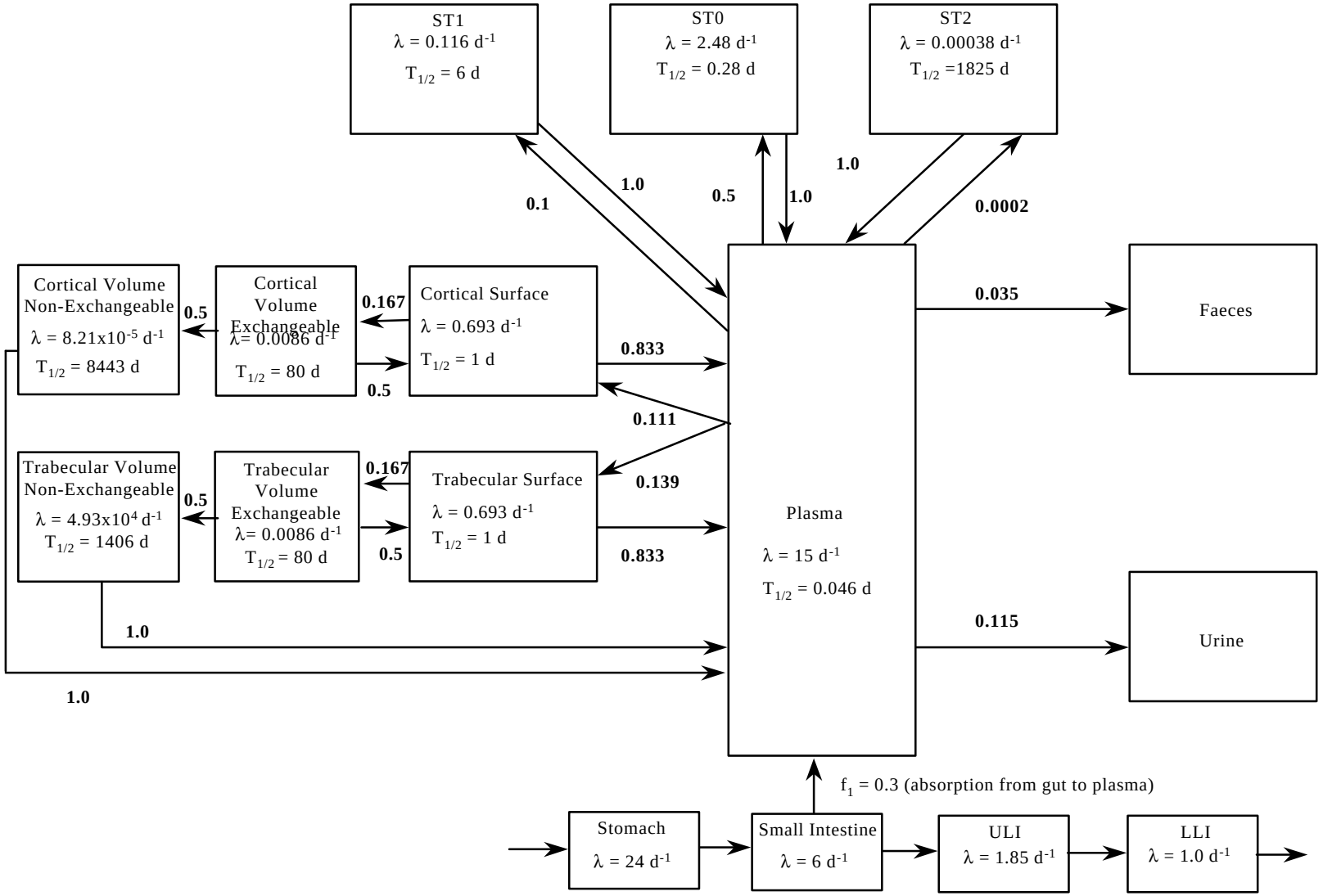


Figure 11.3.1 Metabolic model for Strontium (ICRP, 1993).

Strontium entering the skeleton is initially deposited on the bone surface, but can return to the blood or move to exchangeable bone (cortical and trabecular) volume within a few days (ICRP, 1993) (see Figure 11.3.1). A portion of the activity in the exchangeable bone volume may return to the bone surfaces, while the rest of the activity goes to the nonexchangeable bone volume (cortical and trabecular), where it is gradually removed to the blood (see Figure 11.3.1). Removal from the bone volume occurs by diminution and resorption (Leggett, 1992). Modeling results indicate that 50% of the strontium leaving the exchangeable bone volume goes to the nonexchangeable bone volume (ICRP, 1993). The removal half-time of strontium from the exchangeable bone was modeled to be approximately 80 days (Leggett, 1992).

Information about the flow (or transfer) rates from one compartment of the biokinetic model to another is provided in Table 11.3.2. The publications and data bases reviewed in this work (Leggett, 1992; ICRP, 1989; ICRP, 1979; Schulert et al., 1959) do not provide enough insight for developing probability distributions describing the uncertainty in the transfer rates of the biokinetic model as introduced by natural variability or other sources. To address this issue, Dr. R. Leggett, who is one of the main contributors to the development of the strontium model for the International Commission on Radiological Protection (ICRP), and author of numerous scientific papers in internal dosimetry, was consulted (Leggett, personal communication with C. Lewis, 1997a; 1997b; 1997c).

According to Dr. Leggett, the bone turnover rates are known within a factor of two. The deposition rates from blood to cortical and trabecular surfaces should not vary by more than a factor of 1.5. More measurements are available for the elimination from blood by excretion than for any other transfer rates; therefore, the uncertainty in these elimination rates should not be larger than 50%. The three soft tissue compartments (ST0, ST1, ST2) are empirical kinetic compartments, not physical organs. Less attention was paid by ICRP to these compartments because, in the case of strontium, they have a minimal contribution to the effective dose (which is defined by ICRP as a weighted sum of the doses over all organs). A larger uncertainty (a factor of 3 or 4) was associated with the transfer rates to the soft tissue, because fewer experimental data are available.

The transfer rates between various biokinetic compartments might be correlated. For instance, pregnant women usually have a high absorption of strontium from the gastrointestinal tract to blood that is associated with a high bone turnover rate. Explicit consideration of the correlations should reduce the uncertainty in the dose. However, very little experimental evidence is available about such correlations, and thus they were neglected in this analysis.

Table 11.3.2 Uncertainty associated with the biokinetic parameters for ^{90}Sr.

Path Number	From	To	Flow Rate (days $^{-1}$)	Uncertainty in the Flow Rate	Minimum Value (days $^{-1}$)	Maximum Value (days $^{-1}$)	Distribution
1	Plasma	Cortical Surface	1.7	1.5	1.1	2.5	Loguniform
2	Plasma	Trabecular Surface	2.1	1.5	1.4	3.2	Loguniform
3	Plasma	ST0	7.5	3.5	2.1	26	Loguniform
4	Plasma	ST1	1.5	3.5	0.43	5.3	Loguniform
5	Plasma	ST2	3.0×10^{-3}	3.5	8.6×10^{-4}	0.01	Loguniform
6	Plasma	Feces	0.53	50%	0.26	0.79	Uniform
7	Plasma	Urine	1.7	50%	0.86	2.6	Uniform
8	Cortical Surface	Plasma	0.58	2.0	0.29	1.2	Loguniform
9	Cortical Surface	Exchangeable Cortical Volume	0.12	2.0	0.058	0.23	Loguniform
10	Exchangeable Cortical Volume	Cortical Surface	4.3×10^{-3}	2.0	2.2×10^{-3}	8.6×10^{-3}	Loguniform

Table 11.3.2 (continued)

Path Number	From	To	Flow rate (days $^{-1}$)	Uncertainty in the Flow Rate	Minimum Value (days $^{-1}$)	Maximum Value (days $^{-1}$)	Distribution
11	Exchangeable Cortical Volume	Nonexchangeable Cortical Volume	4.3×10^{-3}	2.0	2.2×10^{-3}	$86. \times 10^{-3}$	Loguniform
12	Nonexchangeable Cortical Volume	Plasma	8.2×10^{-5}	2.0	4.1×10^{-05}	1.6×10^{-4}	Loguniform
13	Trabecular Surface	Plasma	0.58	2.0	2.9×10^{-1}	1.2	Loguniform
14	Trabecular Surface	Exchangeable Trabecular Volume	0.12	2.0	0.06	0.23	Loguniform
15	Exchangeable Trabecular Volume	Trabecular Surface	4.3×10^{-3}	2.0	2.2×10^{-3}	8.6×10^{-3}	Loguniform
16	Exchangeable Trabecular Volume	Nonexchangeable Trabecular Volume	4.3×10^{-3}	2.0	2.2×10^{-3}	8.6×10^{-3}	Loguniform
17	Nonexchangeable Trabecular Volume	Plasma	4.9×10^{-4}	2.0	2.5×10^{-4}	9.9×10^{-4}	Loguniform
18	ST0	Plasma	2.5	3.5	0.71	8.7	Loguniform
19	ST1	Plasma	0.12	3.5	0.033	0.41	Loguniform
20	ST2	Plasma	3.8×10^{-4}	3.5	1.1×10^{-4}	1.3×10^{-3}	Loguniform

In conclusion, an uncertainty factor of 1.5 was assigned to the transfer from blood to bone surfaces, a factor of 2 to all bone exchange rates, and a factor of 3.5 to the transfer between blood and soft tissues. The ICRP default values were divided and multiplied by these factors to obtain the endpoints for loguniform distributions (Table 11.3.2). For the rates of elimination from blood, a 50% uncertainty range around the ICRP default values was used to define uniform distributions. No correlations between the transfer rates were considered in this analysis.

11.3.3.1 Absorption from the Gastrointestinal Tract

Intestinal discrimination between strontium and calcium is essentially a one-step process (Schulert et al., 1959). Strontium that reaches the intestinal tract either passes through the intestinal wall or is eliminated from the body via excretion (Schulert et al., 1959). The fractional absorption (f_1) of strontium from the gastrointestinal tract of normal adults of all mammalian species studied is typically in the range of 0.05 to 0.4; a representative value is probably 0.2 (Coughtrey and Thorne, 1983). Several studies concerning the absorption fraction have been conducted for humans (Underwood, 1977, cited in Coughtrey and Thorne, 1983; Annenkov et al., 1973, cited in Coughtrey and Thorne, 1983). ICRP (1973, cited in Coughtrey and Thorne, 1983) studied fallout data from Britain and New York to establish absorption fractions of 0.198 and 0.18, respectively. Estimates have also been made from fecal excretion. One study indicated that the absorbed fraction was 0.2 for subjects on a normal diet, but the value decreased to approximately 0.15 for those subjects on a high calcium diet (Knizhnikov and Marei, 1967, cited in Coughtrey and Thorne, 1983). Another study was based on the plasma levels of ^{85}Sr and fecal excretion rates (Spencer et al., 1967, cited in Coughtrey and Thorne, 1983). This study indicated a range for the absorption fraction as 0.09-0.35 (Spencer et al., 1967, cited in Coughtrey and Thorne, 1983). Underwood indicated the fractional absorption is typically 0.05 to 0.25 and is consistent with the conclusion of Annenkov et al. (1973) that the fractional absorption is usually between 0.2 and 0.5 (Coughtrey and Thorne, 1983). The authors indicated that the absorption was slightly increased on a low calcium diet and slightly decreased on a high calcium or high phosphorus diet (Spencer et al., 1967, cited in Coughtrey and Thorne, 1983). The absorption for adults can increase if the person is fasting, consuming vitamin D, maintaining a low calcium, magnesium, and phosphorus diet, or living on a milk diet (ICRP, 1993).

A value of 0.3 was recommended by the ICRP (1977; 1993) for adult man for all forms of strontium. The ICRP's recommendation has been endorsed by the Nuclear Energy Agency and the Organization for Economic Cooperation and Development (NEA/OECD) (ICRP, 1993). The proposed value assumes a normal dietary intake of 1 gram of calcium per day (ICRP, 1977; 1993).

To express the uncertainty in the absorption fraction from the gastrointestinal tract (f_1) for ^{90}Sr for the average adult in this study, a triangular distribution was selected. The values selected were 0.1 for the minimum value, 0.2 for the mode, and 0.4 as the maximum value. The range selected and the distribution chosen agree with the values provided in the literature (ICRP, 1979; 1993; Coughtrey and Thorne, 1983).

11.3.3.2 Retention and Elimination of ^{90}Sr

Strontium and calcium behave similarly in the human body. However, strontium is less effectively absorbed from the intestines, more effectively excreted by the kidneys, and less readily incorporated into new bone than calcium (ICRP, 1993). Much of the strontium deposited on the bone surfaces is returned to the plasma within a few days, but a fraction migrates to regions that lose activity to the plasma more slowly (ICRP, 1993). Activity that is returned to the plasma is redistributed or excreted (ICRP, 1993). Kidney discrimination against strontium is a cyclic process in which the strontium, which has been retained in the body during one circulatory cycle due to reabsorption by the renal tubules, is presented to the kidney again during the following circulatory cycle (Schulert et al., 1959). Loss to the urine and feces represents an estimated 15% of strontium atoms leaving the circulation, and the assumed ratio of cumulative urinary to fecal excretion is 3.3 (Leggett, 1992). Activity going to urine is assumed to go directly from plasma to the urinary bladder contents, and activity going to the feces is assumed to go directly from the plasma to the gastrointestinal contents.

11.3.3.3 Special Considerations

Several adjustments to the energy deposition have been made to account for the ^{90}Sr decay in blood while passing through various organs. This type of adjustment is currently used by ICRP in the derivation of the dose factors (Eckerman, 1994). Strontium-90 is primarily a bone seeker; therefore, the transformations occurring in the remaining tissues must be accounted for with respect to the bones. The number of nuclear transformations occurring in the rest of the body (ROB) can be determined by the following.

$$U_{ROB} = U_{ST0} + U_{ST1} + U_{ST2} + \left(\frac{M_{ROB}}{M_{Tbody}} \right) \cdot U_{TC} \qquad \text{(Eq. 11.3.1)}$$

where:

U_{ROB} = the number of nuclear transformations in the rest of the body (unitless);
U_{ST0} = the number of transformations in the compartment with a rapid biological half-life (unitless);
U_{ST1} = the number of transformations in the compartment with a medium biological half-life (unitless);
U_{ST2} = the number of transformations in the compartment with the slowest biological half-life (unitless);
M_{ROB} = the mass of the rest of the body (kg), is determined from the mass of the total body ($M_{T\,body}$) minus the mass of the bones (M_{bone});
$M_{T\,body}$ = the mass of the total body (kg); and
U_{TC} = the number of nuclear transformations in the blood transfer compartment (unitless).

The number of nuclear transformations occurring in the bones is also adjusted for the mass of the remaining tissues. This adjustment is made by using the following equation.

$$U_{bone}^{adj} = U_{bone} + \left(\frac{M_{bone}}{M_{Tbody}} \right) \cdot U_{TC} - \left(\frac{M_{bone}}{M_{ROB}} \right) \cdot U_{ROB} \qquad \text{(Eq. 11.3.2)}$$

where:

U_{bone}^{adj}	=	the adjusted number of nuclear transformations in the bone (unitless).
U_{bone}	=	the number of nuclear transformations in the bone (unitless),
M_{bone}	=	the mass of the bones (kg),
$M_{T\ body}$	=	the mass of the total body (kg),
U_{TC}	=	the number of nuclear transformations in the blood transfer compartment (unitless),
M_{ROB}	=	the mass of the rest of the body (kg), and
U_{ROB}	=	the number of nuclear transformations in the rest of the body (unitless).

The relationships hold for any amount of activity ingested. These adjustments, however, have only a limited influence on the total energy deposited in various organs.

11.3.4 Results and Conclusions

The estimates of the dose factor derived in this study are compared with ICRP (1993) values in Table 11.3.3 and in Figure 11.3.2. Table 11.3.4 provides the 2.5th, 50th, and 97.5th percentiles for the energy per unit intake for all organs. The results provided do not contain the uncertainty introduced by the variability in the organ masses, which is expected to be the most important contributor to the uncertainty. This uncertainty is explicitly taken into consideration when the total dose to a given organ is calculated. The results also do not contain the error in the absorbed fraction (*AF*), which is expected to be low when compared to the other sources of error.

Table 11.3.3 Organ-specific dose conversion factors for adults from ingestion of ^{90}Sr. The ICRP (1993) factors are compared to the distributions derived in this work.

Target Organ	ICRP Reference Man[a] (Sv Bq^{-1})	Present Study (Sv Bq^{-1}) 95% Subjective Confidence Interval		
		lower bound	central value	upper bound
Adrenals	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Bladder	**1.5×10^{-9}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Bone surface	**4.1×10^{-7}**	1.1×10^{-7}	3.2×10^{-7}	7.1×10^{-7}
Brain	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Breast	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Digestive System				
Oral Cavity	**6.7×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Esophagus	**6.7×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Stomach	**9.1×10^{-10}**	3.0×10^{-10}	8.4×10^{-10}	3.0×10^{-9}
Small Intestine	**1.1×10^{-9}**	3.0×10^{-10}	9.7×10^{-10}	3.2×10^{-9}
Upper Large Intestine	**5.9×10^{-9}**	2.1×10^{-9}	4.5×10^{-9}	7.6×10^{-9}
Colon	**2.2×10^{-8}**	1.1×10^{-8}	2.1×10^{-8}	4.9×10^{-8}
Rectum	**2.2×10^{-8}**	1.1×10^{-8}	2.1×10^{-8}	4.9×10^{-8}
Gallbladder	**6.7×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Kidneys	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Liver	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Lungs	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Muscle	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Ovaries	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Pancreas	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Prostate	**6.7×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Red Bone Marrow	**1.8×10^{-7}**	2.3×10^{-8}	1.4×10^{-7}	4.2×10^{-7}
Skin	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Spleen	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Testes	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Thymus	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Thyroid	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Uterus	**6.6×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Remainder[b]	**6.7×10^{-10}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Effective (ICRP)[c]	**2.8×10^{-8}**	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}

[a]ICRP Publication 67 (ICRP,1993).

[b]Based on the value for remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

[c]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0.

Table 11.3.4 Energy deposited in various organs from ingestion of 1 Bq of ^{90}Sr by adults.

Target Organ	Energy Deposited in Each Target from an Unit Intake [J/Bq]		
	95% Subjective Confidence Interval		
	lower bound	central value	upper bound
Adrenals	1.5×10^{-12}	8.8×10^{-12}	4.0×10^{-11}
Bladder	5.0×10^{-12}	2.8×10^{-11}	1.3×10^{-10}
Bone surface	1.3×10^{-8}	3.9×10^{-8}	8.5×10^{-8}
Brain	1.5×10^{-10}	8.8×10^{-10}	4.0×10^{-9}
Breast	4.0×10^{-11}	2.3×10^{-10}	1.0×10^{-9}
Digestive System			
Oral Cavity	1.1×10^{-11}	6.3×10^{-11}	2.9×10^{-10}
Esophagus	4.1×10^{-12}	2.3×10^{-11}	1.1×10^{-10}
Stomach	7.5×10^{-11}	2.1×10^{-10}	7.5×10^{-10}
Small intestine	1.2×10^{-10}	3.9×10^{-10}	1.3×10^{-9}
Upper Large Intestine	4.7×10^{-10}	9.8×10^{-10}	1.7×10^{-9}
Lower Large Intestine	1.5×10^{-9}	2.8×10^{-9}	6.7×10^{-9}
Rectum	4.0×10^{-10}	7.4×10^{-10}	1.7×10^{-9}
Gallbladder	1.1×10^{-12}	6.3×10^{-12}	2.9×10^{-11}
Kidneys	3.4×10^{-11}	2.0×10^{-10}	8.9×10^{-10}
Liver	2.0×10^{-10}	1.1×10^{-9}	5.2×10^{-9}
Lungs	1.1×10^{-10}	6.3×10^{-10}	2.9×10^{-9}
Muscle	3.1×10^{-9}	1.8×10^{-8}	8.1×10^{-8}
Ovaries	1.2×10^{-12}	6.9×10^{-12}	3.2×10^{-11}
Pancreas	1.1×10^{-11}	6.3×10^{-11}	2.9×10^{-10}
Prostate	1.8×10^{-12}	1.8×10^{-11}	4.6×10^{-11}
Red Bone Marrow	3.5×10^{-8}	2.1×10^{-7}	6.3×10^{-7}
Skin	2.9×10^{-10}	1.6×10^{-9}	7.5×10^{-9}
Spleen	2.0×10^{-11}	1.1×10^{-10}	5.2×10^{-10}
Testes	3.9×10^{-12}	2.2×10^{-11}	1.0×10^{-10}
Thymus	2.2×10^{-12}	1.3×10^{-11}	5.8×10^{-11}
Thyroid	2.2×10^{-12}	1.3×10^{-11}	5.8×10^{-11}
Uterus	8.8×10^{-12}	5.1×10^{-11}	2.3×10^{-10}
Remainder[a]	6.4×10^{-9}	3.7×10^{-8}	1.7×10^{-7}
Effective (ICRP)[b]	7.6×10^{-9}	4.3×10^{-8}	2.0×10^{-7}

[a]Based on the value for remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

[b]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0.

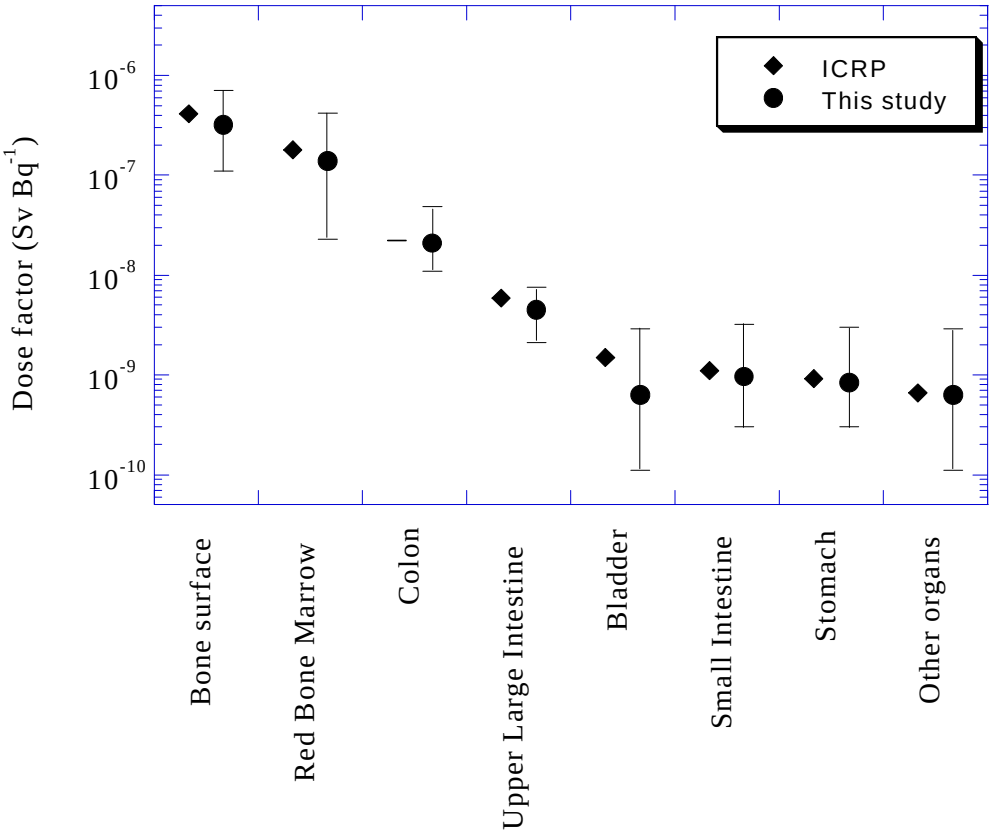


Figure 11.3.2 Comparison of the ingestion dose factors for $^{90}Sr/^{90}Y$ calculated in this study to the ICRP (1993) values

ICRP (1993) provides an effective dose factor, which is indicative of a whole body dose, and organ-specific dose factors. Using the ICRP (1993) values, the organs contributing most to the effective dose factor for strontium are the bones, red bone marrow, lower large intestine, upper large intestine, bladder, small intestine, and stomach, respectively. In this study, the organs contributing the most to the effective dose factor were the bone surface, red bone marrow, lower large intestine, and upper large intestine, respectively. The uncertainty factors associated with the digestive organs were 3.5, 3.3, 1.7, and 2.0 for the stomach, small intestine, upper large intestine, and lower large intestine, respectively. The uncertainty associated with the bone dose factor was a factor of 2.2, and that of the red bone marrow was 2.9.

11.3.5 Sensitivity Analysis

A sensitivity analysis was performed to determine the most important contributors to the uncertainty in the energy deposited in each organ of interest from a unit ingestion of ^{90}Sr. The energy deposited in the gastrointestinal tract organs is given by the amount of ^{90}Sr present in the content and in the walls of these organs. The energy delivered by ^{90}Sr located in the walls is comparable to the energy delivered by the ^{90}Sr in the organ contents. The number of disintegrations in the organ walls is obtained similarly to any other soft tissues; therefore, the transfer rates between plasma and soft tissues influence the dose to the organ walls. For stomach and small intestine, the transfer rates between the medium turnover soft tissue (ST1) and plasma are the most important contributors to the uncertainty. On the other hand, the elimination rate from the upper large intestine and from colon are the most important contributors for these two organs. A complete description of the dependency and of the actual contributions to the uncertainties is presented in Table 11.3.5. A selection of the parameters that contributed more than 5% is presented in Figures 11.3.3 and 11.3.4.

For irradiation of the bone and of different soft tissues, each of the biokinetic model parameters was included in the sensitivity analysis (Table 11.3.5). The contributions of various parameters is fairly uniform for the all organs other than the components of the gastrointestinal tract.

The uncertainty in the energy deposited in the bone surface is dominated by the transfer rate from the cortical bone surface to the blood, while the rest of the transfer rates have very close values. Since the sensitivity analysis is based on Spearman's rank correlation, the coefficient is only an approximating method for variance decomposition.

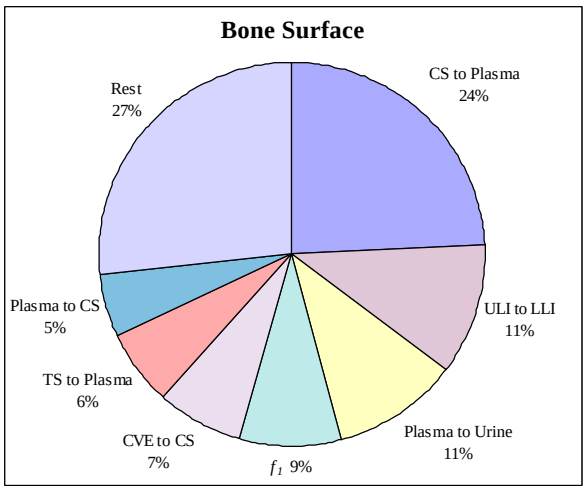
Table 11.3.5 Contribution to the uncertainty in the dose factor (%) of different transfer rates between the compartments of the biokinetic model[a].

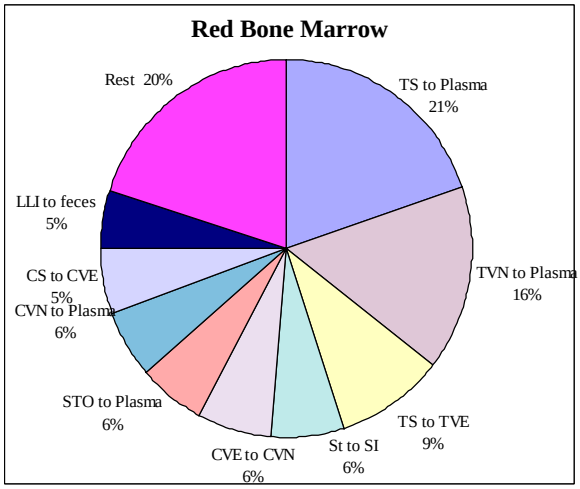
Transfer Rate	Bone Surface	Red Bone Marrow	Soft Tissues	Stomach	Small Intestine	Upper Large Intestine	Lower Large Intestine
Plasma to Cortical Surface	5.3	0.3	4.5	3.6	3.7	8.6	8.3
Plasma to Trabecular Surface	0.3	1.0	0.2	0.2	1.0	0.0	0.2
Plasma to Soft Tissue (fast - ST0)	2.0	0.7	1.4	1.2	2.7	0.0	0.0
Plasma to Soft Tissue (medium - ST1)	1.0	3.7	**15.9**	**17.3**	**16.7**	7.3	1.0
Plasma to Soft Tissue (slow - ST2)	3.3	0.8	1.3	0.7	1.2	0.1	0.2
Plasma to Feces	0.8	1.2	6.5	6.6	5.9	5.9	0.1
Plasma to Urine	**10.8**	0.5	**10.3**	**11.4**	**12.2**	**10.2**	0.1
Cortical Surface to Plasma	**24.1**	2.1	0.2	0.2	0.1	1.9	0.0
Cortical Surface to Cortical Volume, Exchangeable	1.7	5.3	0.3	0.1	0.8	0.3	0.2
Cortical Volume Exchangeable to Cortical Surface	7.1	0.3	0.1	0.2	0.0	1.2	0.5
Cortical Volume Exchangeable to Cortical Volume, Nonexchangeable	0.1	6.2	2.6	1.8	0.6	0.5	2.0
Cortical Volume Nonexchangeable to Plasma	0.0	5.9	0.1	0.1	0.0	0.4	2.5
Trabecular Surface to Plasma	6.4	**19.8**	4.3	4.9	2.6	0.5	6.1
Trabecular Surface to TVE	1.7	9.3	6.1	5.2	7.1	0.7	0.1
Trabecular Volume Exchangeable to TS	0.7	0.2	0.7	0.8	1.5	0.9	0.1
Trabecular Volume Exchangable to TVN	2.4	2.0	0.2	0.1	0.1	0.1	0.8
Trabecular Volume Nonexchangeable to Plasma	4.5	**15.9**	0.5	0.0	0.0	0.2	4.4
Soft Tissue (fast - ST0) to Plasma	1.8	6.0	0.4	1.0	1.1	0.6	3.5
Soft Tissue (medium - ST1) to Plasma	0.0	1.9	**32.9**	**34.7**	**27.2**	9.8	0.0

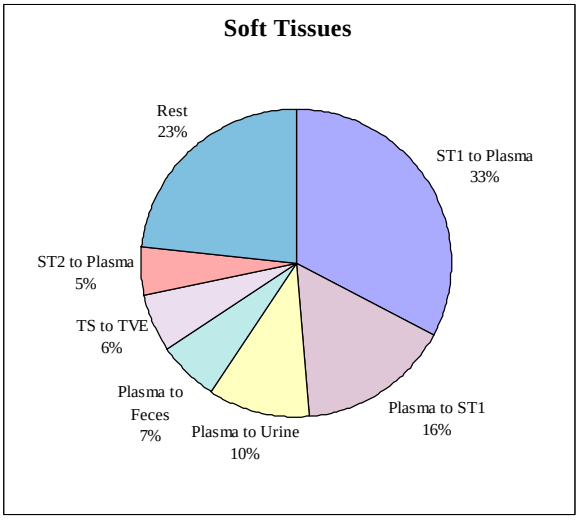
Table 11.3.5 (continued)

Transfer Rate	Bone Surface	Red Bone Marrow	Soft Tissues	Stomach	Small Intestine	Upper Large Intestine	Lower Large Intestine
Soft Tissue (slow - ST2) to Plasma	0.8	0.7	5.0	5.6	3.4	0.3	0.3
Stomach to Small Intestine	0.2	6.4	2.1	0.1	1.3	0.7	0.9
Small Intestine to Upper Large Intestine and Plasma	2.2	1.3	0.1	0.1	7.7	3.7	7.7
Upper Large Intestine to Lower Large Intestine	11.1	0.7	2.0	1.9	2.5	**41.9**	6.7
Lower Large Intestine to Feces	3.2	5.0	0.0	0.0	0.3	1.2	**50.0**
Absorption from the gut (f_1)	8.6	2.7	2.4	2.3	0.4	3.0	4.4

a The most important contributions to the uncertainty are shown in bold face type.



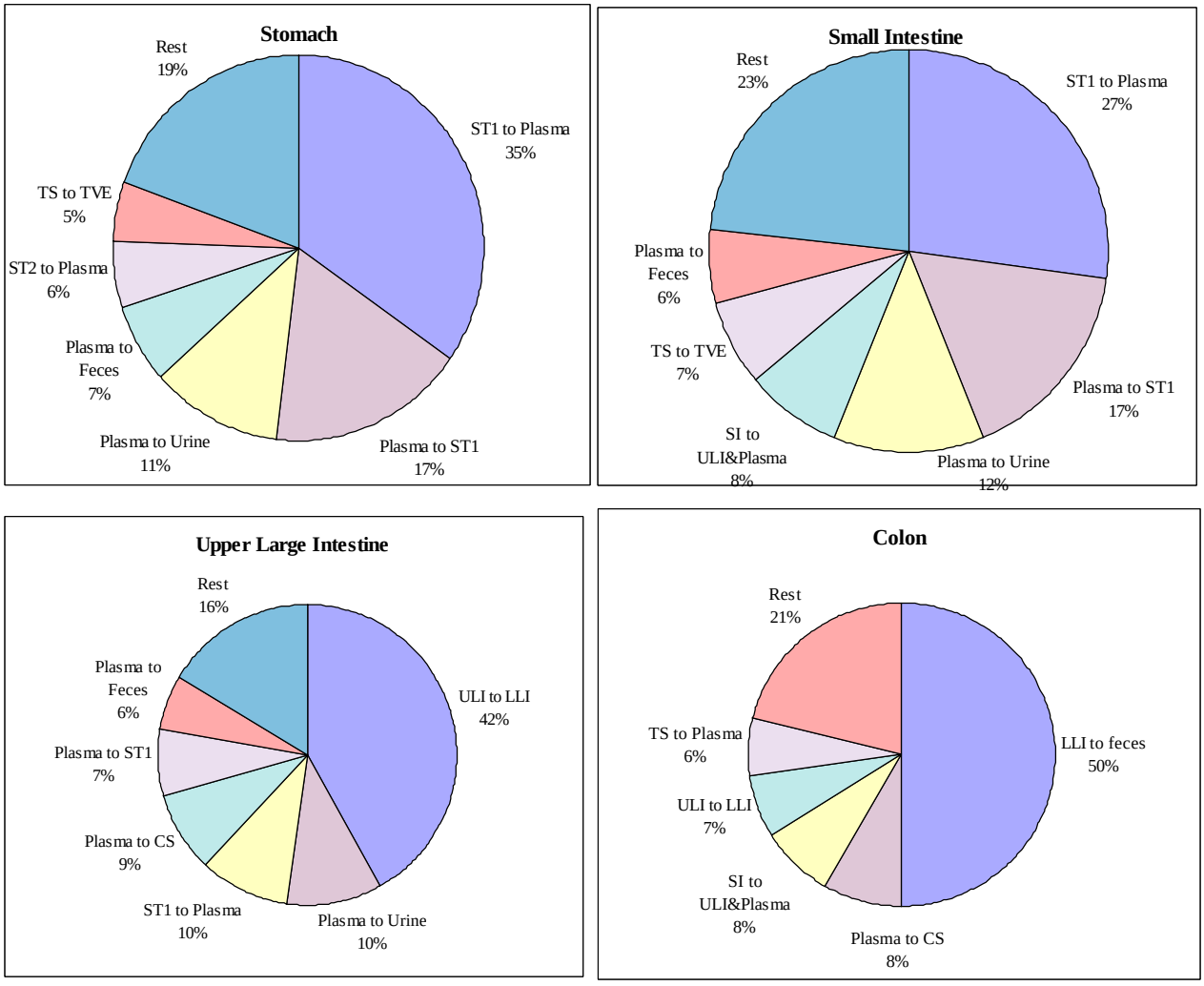




LEGEND:

CS = Cortical Surface
TS = Trabecular Surface
CVE = Cortical Volume Exchangeable
TVE = Trabecular Volume Exchangeable
CVN = Cortical Volume Non-Excheangeable
TVN = Trabecular Volume Non-Exchangeable
ST0 = Soft Tissue (fast turnover)
ST1 = Soft Tissue (medium turnover)
ST2 = Soft Tissue (slow turnover)
SI = Small Intestine
f_1 = Absorption fraction from intestines to blood

Figure 11.3.3 The results of the sensitivity analysis for the ^{90}Sr dose factors for bone surface, red bone marrow, and soft tissues.



LEGEND

CS	= Cortical Surface	ST0	= Soft Tissue (fast turnover)
TS	= Trabecular Surface	ST1	= Soft Tissue (medium turnover)
CVE	= Cortical Volume Exchangeable	ST2	= Soft Tissue (slow turnover)
TVE	= Trabecular Volume Exchangeable	SI	= Small Intestine
CVN	= Cortical Volume Non-Excheangeable	ULI	= Upper Large Intestine
TVN	= Trabecular Volume Non-Exchangeable	LLI	= Lower Large Intestine

Figure 11.3.4 The results of the sensitivity analysis for the ^{90}Sr dose factors for the gastrointestinal tract organs.

It is difficult to distinguish between the rest of the contributors to the bone surface's energy deposition because the correlation coefficients are very similar.

For red bone marrow, the most important contributors to the uncertainty in the deposited energy are the trabecular bone surface transfer rate to blood and the trabecular bone volume transfer rate to blood, followed by the transfer rate of ^{90}Sr from the trabecular bone surface to the exchangeable trabecular bone volume. The rest of the important parameters have such similar correlation coefficient values that an attempt to rank them was not warranted.

The ^{90}Sr exchange rates between blood and the soft tissue having a medium biological half-life (ST1 in Figure 11.3.1, with $T_{1/2}$ = 6 days) are the most important contributors to the uncertainty. The accumulation of ^{90}Sr in the rest of the soft tissue compartments (ST0 and ST2) has a much lower contribution, because the fraction of ^{90}Sr transferred from the blood to ST2 is low compared with the fraction transferred to ST0 and ST1 (Figure 11.3.1). In addition, the turnover time in ST0 is rapid, which decreases the amount of time available for energy deposition. The next largest contribution to the uncertainty is the transfer rate of ^{90}Sr from the blood to excretion.

11.4 Cobalt-60

Cobalt-60 was determined during the initial screening analysis to be an important radioactive element released from the Oak Ridge Reservation via White Oak Creek. As a result, the dose contribution of ^{60}Co to the overall dose possibly received by target individuals living along the Clinch River was required. The dose factor (DF) is one element of uncertainty in the dose. The purpose of this section is to assess the uncertainty in the DF, to determine the most important factors contributing to the uncertainty in the DF for ^{60}Co (sensitivity analysis), and to describe the way in which this uncertainty will be treated in the risk assessment.

11.4.1 Background Information

Cobalt has one isotope found in nature (^{59}Co), while the remaining isotopes are man-made. Cobalt-60 is used as a gamma source for various applications including radiotherapy (Weast, 1968). Nonradioactive cobalt appears to be essential to life, since it is incorporated into vitamin B_{12}. Natural cobalt is present in trace quantities in all foods, but is richest in green leafy vegetables. According to ICRP (1975), humans have an intake of about 300 μg of cobalt per day. Cobalt is eliminated in urine (about two thirds) and feces (one third) (ICRP, 1975).

11.4.2 Absorbed Fraction

The majority of the energy deposited in various organs comes from the two gamma rays emitted by ^{60}Co. Even for the digestive system organs, the contribution of gamma rays from cobalt stored for a long term in all tissues should be larger than the contribution of the decays during the relative short time of passage

through the gastrointestinal tract. As was shown in Section 11.2.6, for a gamma emitter that distributes fairly uniformly in the body, the contribution of the uncertainty in the absorbed fraction (*AF*) is small and can be neglected. Thus, as with ^{137}Cs, the uncertainty associated with the absorbed fraction (*AF*) for ^{60}Co was neglected.

11.4.3 Biokinetics of ^{60}Co in the Human Body

A number of studies have been performed using radioactive cobalt in humans (Engel et al., 1967; Paley and Sussman, 1963; Letourneau et al., 1972; Smith et al., 1972; Miltenberger et al., 1981). Among the animal experiments performed, the majority have involved rats (Taylor, 1961; Nishimura et al., 1976; Hollins and McCullough, 1971; Inaba et al., 1982). However, mice, dogs and monkeys have also been utilized (Thomas et al., 1976). In Thomas et al., significant differences were found between animal data and human data (Table 11.4.1), and the authors suggest that animal data can not safely be used for human dosimetry. Therefore, quantitative measurements of cobalt in human subjects are the most relevant sources of information and were used for this analysis.

11.4.3.1 Absorption from the Gastrointestinal Tract

ICRP (1993) recommends an absorption fraction (f_1) of 10% for adults. The fraction of radioactive cobalt absorbed from the gastrointestinal tract to the blood measured by Smith et al. (1972) varied between 1% and 5% for normal subjects and between 2% and 17% if stable cobalt was added. In fasted patients, 3% - 5% was observed in normal patients and 20% -28% absorption was observed in patients who had been administered stable cobalt as well. Paley and Sussman (1963) obtained an absorption fraction of about 3% for 37 human subjects. A piecewise uniform distribution having a mean of 10%, a lower value of 2% and an upper value of 20%, was used to express the uncertainty in the fraction of cobalt absorbed from the gastrointestinal tract. This distribution was then normalized to the mean value of 10%. The result of the normalization is a distribution centered on 1, and a range dictated by the uncertainty in the absorption fraction (f_1). The normalization was performed to obtain a multiplicative correction factor that will be applied to the dose conversion factor. This factor describes the uncertainty associated with the transfer from the gastrointestinal tract to blood.

Table 11.4.1 The whole-body retention function for cobalt from different experiments after intravenous injection. The retention function is expressed as a sum of decaying exponential terms, weighted by the coefficients a_i. The table lists the biological half-lives (days) of each exponential term (Section 11.4.3.2).

Subject	Compound	Reference	a_1	$T_{1/2}$ (1)	a_2	$T_{1/2}$ (2)	a_3	$T_{1/2}$ (3)	a_4	$T_{1/2}$ (4)	a_5	$T_{1/2}$ (5)
1	$^{58}CoCl_2$	A	0.24	0.033	0.36	1.75	0.19	8.9	0.15	67.9	0.06	0
2	$^{58}CoCl_2$	A	0.30	0.007	0.36	2.92	0.06	10.5	0.16	26.9	0.12	481
3	$^{58}CoCl_2$	A	0.26	0.011	0.34	2.01	0.17	9.6	0.11	34.5	0.13	265
4	$^{58}CoCl_2$	A	0.31	0.011	0.26	1.30	0.21	7.4	0.16	59.7	0.06	1690
5	$^{58}CoCl_2$	A	0.44	0.277	0.22	1.56	0.18	7.7	0.13	65.4	0.04	0
6	$^{58}CoCl_2$	A	0.31	0.007	0.32	1.33	0.14	7.6	0.13	49.5	0.09	1155
7	$^{58}CoCl_2$	A	0.47	0.063	0.22	1.56	0.16	8.5	0.10	68.6	0.05	0
8	$^{58}CoCl_2$	A	0.28	0.099	0.28	1.41	0.18	8.4	0.17	59.7	0.09	0
9	$^{58}CoCl_2$	A	0.20	0.018	0.38	1.22	0.20	9.1	0.11	45.3	0.11	546
10	$^{58}CoCl_2$	A	0.41	0.347	0.22	1.72	0.18	9.6	0.13	75.3	0.05	0
11	$^{58}CoCl_2$	A	0.22	0.023	0.34	1.13	0.18	4.9	0.14	25.7	0.12	261
12	$^{58}CoCl_2$	A	0.22	0.173	0.31	1.12	0.21	7.8	0.15	34.1	0.11	436
13	$^{58}CoCl_2$	A	0.22	0.021	0.42	2.86	0.16	15.6	0.11	67.9	0.09	729
14	$^{58}CoCl_2$	A	0.41	0.408	0.27	5.02	0.16	30.4	0.10	126.0	0.07	0
15	$^{58}CoCl_2$	A	0.44	0.301	0.19	2.63	0.18	10.7	0.13	64.2	0.06	0
16	$^{58}CoCl_2$	A	0.45	0.365	0.22	2.97	0.18	13.1	0.11	77.9	0.05	0
Collective fit	$^{58}CoCl_2$	A	0.36	0.25	0.24	1.78	0.19	8.1	0.13	47.5	0.09	608
1	$^{60}CoCl_2$	B	0.33	0.37	--	--	0.36	3.6	0.18	45	0.13	830
2	$^{60}CoCl_2$	B	0.55	0.71	--	--	0.28	7.6	0.08	81	0.09	770

Table 11.4.1 (continued)

Subject	Compound	Reference	a_1	$T_{1/2}$ (1)	a_2	$T_{1/2}$ (2)	a_3	$T_{1/2}$ (3)	a_4	$T_{1/2}$ (4)	a_5	$T_{1/2}$ (5)
ICRP	any	C	0.5	0.5	--	--	0.3	6	0.1	60	0.1	800
Mouse	$^{60}CoCl_2$	D	0.87	0.60	--	--	0.10	5.78	0.02	49.5	0.0086	495
Rat	$^{60}CoCl_2$	D	0.90	0.45	--	--	0.07	5.33	0.02	31.5	0.0074	315
Monkey	$^{60}CoCl_2$	D	0.84	0.56	--	--	0.07	6.93	0.06	23.9	0.03	182
Dog	$^{60}CoCl_2$	D	0.81	0.73	--	--	0.10	6.93	0.05	31.5	0.041	178

[A] Letourneau et al., 1972
[B] Smith et al., 1972
[C] ICRP, 1989
[D] Thomas et al., 1976

11.4.3.2 Retention and Elimination of ^{60}Co

The liver, the kidney, and the urinary bladder tend to accumulate more cobalt within a short time after ingestion or intravenous injection than the other organs. During longer periods of time after administration, cobalt is uniformly distributed in the human body, with the exception of the liver (elevated levels after 1,000 days in human subjects) (Leggett, 1997a, b). ICRP (1989) uses a biokinetic model containing seven compartments: blood, three other tissue compartments (non-liver) and three liver compartments (Figure 11.4.1). These compartments do not necessarily refer to physiological compartments, but are based on data from different human studies. The liver as a whole and the tissue compartments have similar biological half-lives, and thus the cobalt retention function[4] can be represented as a sum of four decaying exponential terms.

Letourneau et al. (1972) studied the metabolism of cobalt in normal human males. Sixteen subjects were injected with $^{58}CoCl_2$ and followed for 312 to 382 days. The amount of cobalt in the body was measured, and different retention functions were fit to the data. The best fit was obtained for a sum of five exponential terms (Figure 11.4.2). The doses per unit intake have been calculated for all 16 subjects. The uncertainty of the doses is given only by the uncertainty in the biokinetic parameters.

Smith et al. (1972) studied the retention of cobalt in men. Eleven male subjects were injected intravenously with $^{60}CoCl_2$, and two of them were followed for 1018 days. The best fit of the retention function was obtained for a sum of four exponential terms (Table 11.4.1, Figure 11.4.2).
Finally, Miltenberger et al. (1981) investigated the ^{60}Co body burden of the Marshallese population exposed to nuclear weapons test fallout. From available data, the number of compartments that should be used in the ^{60}Co retention model could not be estimated, and only an estimate of the long-term biological removal rate constant was derived. Since limited information was obtained from this study, it was not used as a basis for this assessment.

[4]A retention function is represented by the following expression: $R(t) = \sum_{i=1}^{\frac{N}{2}} a_i \cdot e^{-b_i t}$ where N is the number of exponential terms.

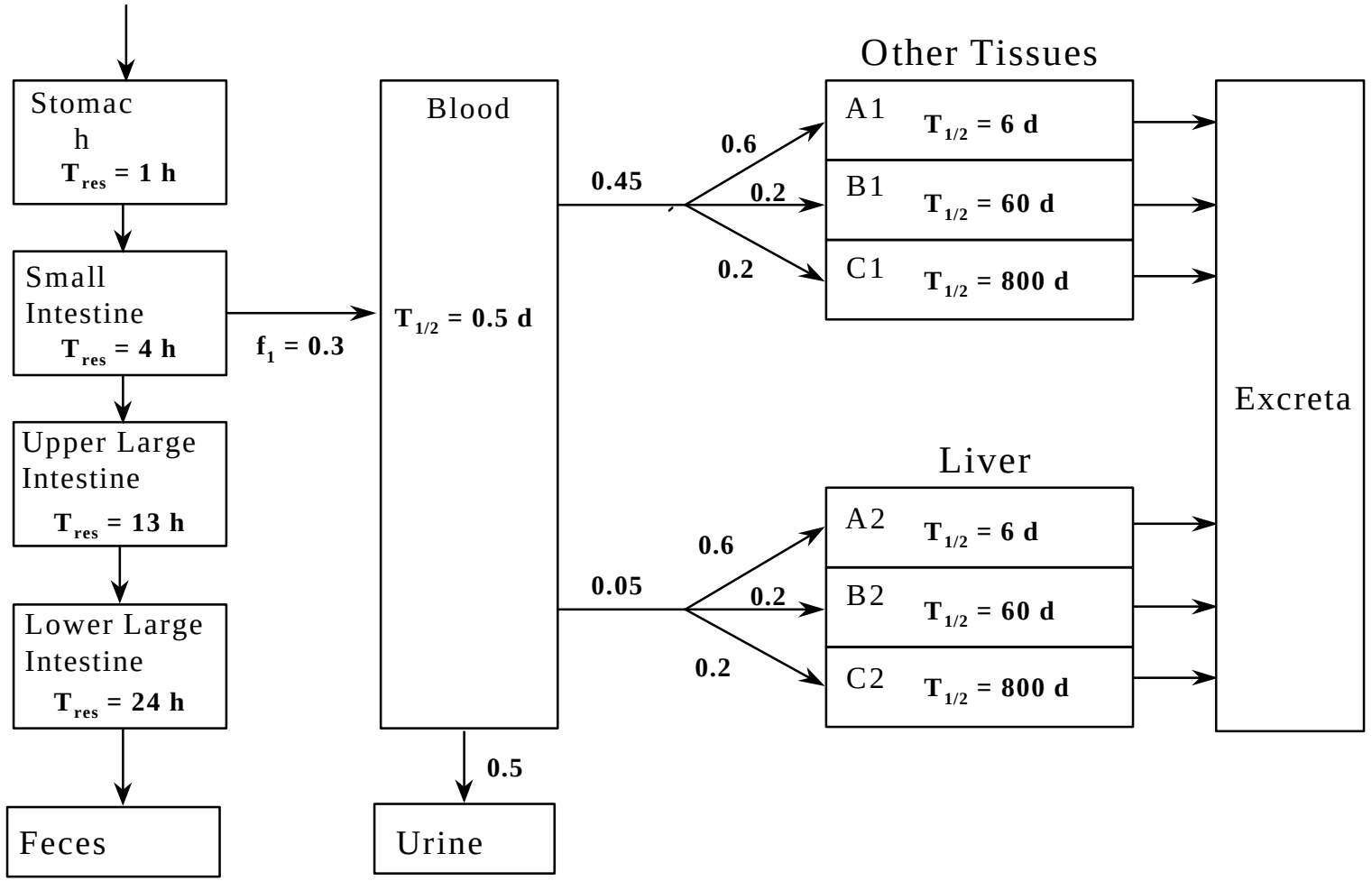


Figure 11.4.1 ICRP metabolic model for ^{60}Co (ICRP, 1977).

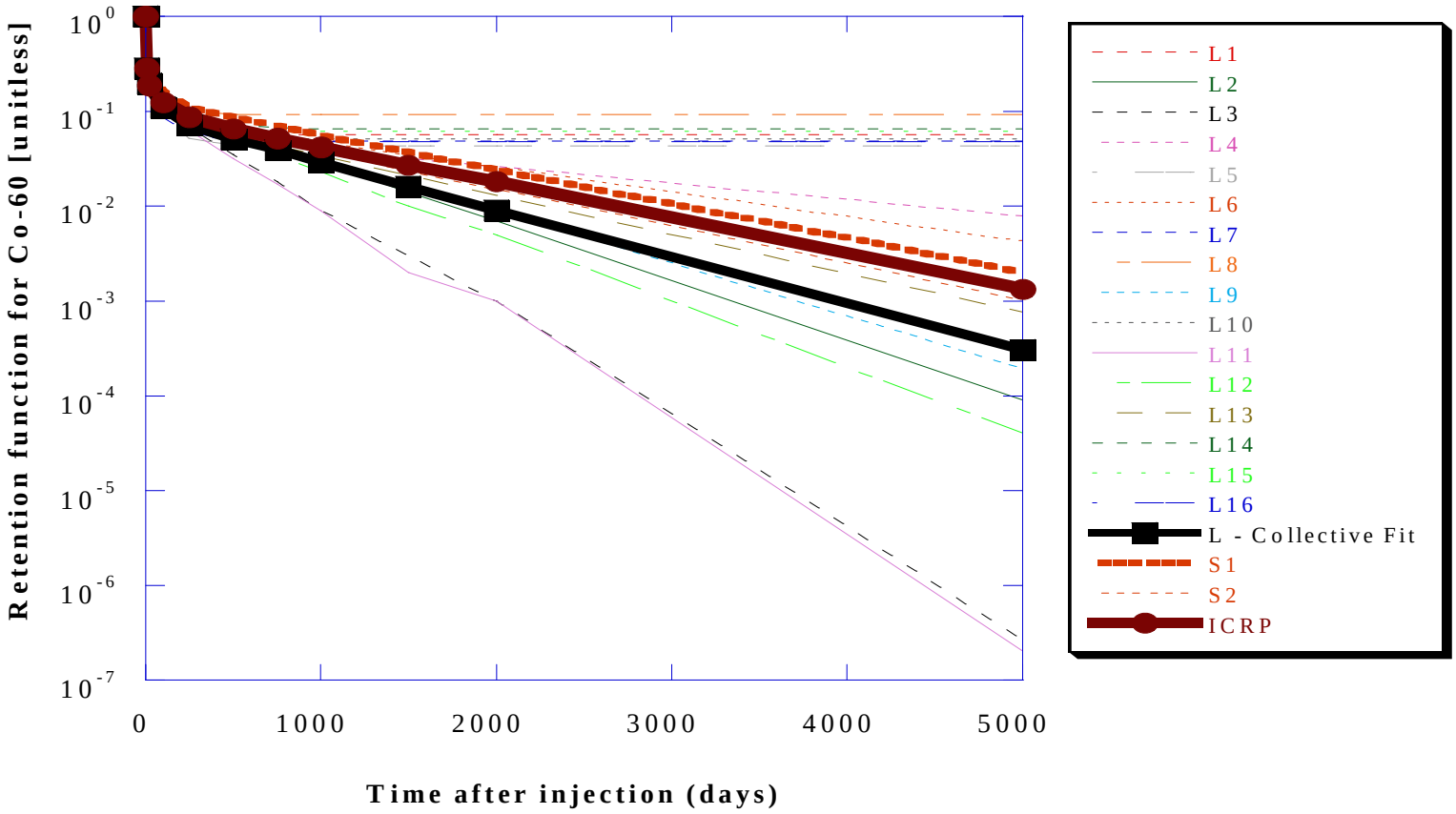


Figure 11.4.2 Variability of cobalt retention in humans after intravenous injection. Data from Letourneau et al. (1972) (L), and Smith et al. (1972) (S) are compared to the ICRP (1993) retention function.

The biokinetics of cobalt after an intravenous administration can be expressed as a sum of exponential terms. The retention function can be integrated to obtain the number of nuclear transformations in the whole body. This number is not representative for ingestion, since cobalt is only partially absorbed from the gastrointestinal tract. However, the uncertainty in the above number of transformations gives an insight on the uncertainty in the biokinetics in general. A summary of the estimated number of nuclear transformations is presented in Table 11.4.2. For human subjects exposed in the studies of Letourneau et al. (1972) and Smith et al. (1972), all of the experimental values are found within a range not exceeding a factor of 5 around the average value. The factor of five was determined by taking the ratios between the different observed values (average/minimum, and maximum/average). Therefore, an uncertainty factor of five is thought to be representative of the uncertainty introduced by the biokinetics of cobalt. A piecewise uniform distribution having a mean value of 1 and a range from 0.2 to 5 (a factor of 5) was designed to represent this uncertainty. This distribution was centered on the mean value rather than on the median.

11.4.4 Results and Conclusions

In this study, the dose factors for ^{60}Co were not recalculated, but only an uncertainty factor was estimated and then applied to the ICRP dose factors. The uncertainty factor was obtained by combining the uncertainty introduced by the absorption fraction (f_1) from the gastrointestinal tract with the uncertainty introduced by the biokinetics of cobalt once it is present in blood. A total uncertainty factor was estimated by multiplying the two individual uncertainty factors, and then applying them to all organs. An uncertainty factor of about 9.7 (measured as the ratio between the 97.5%-tile and the 50%-tile) was obtained for dose conversion factors (Table 11.4.3, Figure 11.4.3). The estimated energy per unit intake and the associated uncertainties are presented in Table 11.4.4.

Table 11.4.2 Summary of the estimated number of nuclear transformations based on the experimentally determined retention in humans after intravenous administration of radioactive cobalt.

Number of Nuclear Transformations	
$^{58}CoCl_2$ data[a]	
Average	5.12×10^6
Standard deviation	4.43×10^6
Minimum	1.09×10^6
Median	3.47×10^6
Maximum	1.40×10^7
$^{60}CoCl_2$ data[b]	
Average	1.22×10^7
Minimum	9.76×10^6
Maximum	1.46×10^7
ICRP[c]	1.10×10^7

[a] 11 subjects followed-up by Letourneau et al. (1972).
[b] 2 subjects followed-up by Smith et al. (1972).
[c] ICRP default value.

Table 11.4.3 Organ-specific dose conversion factors for adults from ingestion of ^{60}Co. The ICRP default factors are compared to the distributions derived in this work.

Target Organ	ICRP Reference Man[a] (Sv Bq^{-1})	Present Study (Sv Bq^{-1}) 95% Subjective Confidence Interval		
		lower bound	central value	upper bound
Adrenals	**2.5×10^{-9}**	2.8×10^{-10}	1.5×10^{-9}	1.4×10^{-8}
Bladder	**2.6×10^{-9}**	2.9×10^{-10}	1.5×10^{-9}	1.5×10^{-8}
Bone surface	**2.1×10^{-9}**	2.4×10^{-10}	1.2×10^{-9}	1.2×10^{-8}
Brain	**1.4×10^{-9}**	1.6×10^{-10}	8.2×10^{-10}	7.9×10^{-9}
Breast	**1.4×10^{-9}**	1.6×10^{-10}	8.2×10^{-10}	7.9×10^{-9}
Digestive System				
Oral Cavity	**2.0×10^{-9}**	2.3×10^{-10}	1.2×10^{-9}	1.1×10^{-8}
Esophagus	**2.0×10^{-9}**	2.3×10^{-10}	1.2×10^{-9}	1.1×10^{-8}
Stomach	**2.6×10^{-9}**	2.9×10^{-10}	1.5×10^{-9}	1.5×10^{-8}
Small Intestine	**4.2×10^{-9}**	4.8×10^{-10}	2.4×10^{-9}	2.4×10^{-8}
Upper Large Intestine	**6.6×10^{-9}**	7.5×10^{-10}	3.8×10^{-9}	3.7×10^{-8}
Colon	**1.2×10^{-8}**	1.4×10^{-9}	7.0×10^{-9}	6.8×10^{-8}
Rectum	**1.2×10^{-8}**	1.4×10^{-9}	7.0×10^{-9}	6.8×10^{-8}
Gallbladder	**2.0×10^{-9}**	2.3×10^{-10}	1.2×10^{-9}	1.1×10^{-8}
Kidneys	**2.4×10^{-9}**	2.7×10^{-10}	1.4×10^{-9}	1.4×10^{-8}
Liver	**4.5×10^{-9}**	5.1×10^{-10}	2.6×10^{-9}	2.5×10^{-8}
Lungs	**1.8×10^{-9}**	2.0×10^{-10}	1.0×10^{-9}	1.0×10^{-8}
Muscle	**1.9×10^{-9}**	2.2×10^{-10}	1.1×10^{-9}	1.1×10^{-8}
Ovaries	**4.3×10^{-9}**	4.9×10^{-10}	2.5×10^{-9}	2.4×10^{-8}
Pancreas	**2.6×10^{-9}**	2.9×10^{-10}	1.5×10^{-9}	1.5×10^{-8}
Prostate	**2.0×10^{-9}**	2.3×10^{-10}	1.2×10^{-9}	1.1×10^{-8}
Red Bone Marrow	**2.2×10^{-9}**	2.5×10^{-10}	1.3×10^{-9}	1.2×10^{-8}
Skin	**1.3×10^{-9}**	1.5×10^{-10}	7.6×10^{-10}	7.3×10^{-9}
Spleen	**2.1×10^{-9}**	2.4×10^{-10}	1.2×10^{-9}	1.2×10^{-8}
Testes	**1.8×10^{-9}**	2.0×10^{-10}	1.0×10^{-9}	1.0×10^{-8}
Thymus	**1.7×10^{-9}**	1.9×10^{-10}	9.9×10^{-10}	9.6×10^{-9}
Thyroid	**1.7×10^{-9}**	1.9×10^{-10}	9.9×10^{-10}	9.6×10^{-9}
Uterus	**3.1×10^{-9}**	3.5×10^{-10}	1.8×10^{-9}	1.7×10^{-8}
Remainder[b]	**2.0×10^{-9}**	2.3×10^{-10}	1.2×10^{-9}	1.1×10^{-8}
Effective (ICRP)[c]	**3.4×10^{-9}**	3.8×10^{-10}	2.0×10^{-9}	1.9×10^{-8}

[a]ICRP Publication 67 (ICRP,1993).

[b]Based on the value for remainder as reported by ICRP Publication 67 (ICRP, 1993). The "remainder" is defined in Section 11.0.

[c]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0.

Table 11.4.4 - Energy deposited in various organs from ingestion of 1 Bq of ^{60}Co by adults.

Target Organ	Energy deposited in each target from an unit intake [J/Bq]		
	95% subjective confidence interval		
	lower bound	central value	upper bound
Adrenals	4.0×10^{-12}	2.0×10^{-11}	2.0×10^{-10}
Bladder	1.3×10^{-11}	6.8×10^{-11}	6.6×10^{-10}
Bone surface	2.9×10^{-11}	1.5×10^{-10}	1.4×10^{-9}
Brain	2.2×10^{-10}	1.1×10^{-9}	1.1×10^{-8}
Breast	5.7×10^{-11}	2.9×10^{-10}	2.8×10^{-9}
Digestive System			
Oral Cavity	2.3×10^{-11}	1.2×10^{-10}	1.1×10^{-9}
Esophagus	8.4×10^{-12}	4.3×10^{-11}	4.2×10^{-10}
Stomach	4.4×10^{-11}	2.3×10^{-10}	2.2×10^{-9}
Small intestine	3.0×10^{-10}	1.6×10^{-9}	1.5×10^{-8}
Upper Large Intestine	1.6×10^{-10}	8.1×10^{-10}	7.8×10^{-9}
Lower Large Intestine	2.2×10^{-11}	1.1×10^{-10}	1.1×10^{-9}
Rectum	4.8×10^{-11}	2.4×10^{-10}	2.4×10^{-9}
Gallbladder	2.3×10^{-12}	1.2×10^{-11}	1.1×10^{-10}
Kidneys	8.4×10^{-11}	4.3×10^{-10}	4.2×10^{-9}
Liver	9.2×10^{-10}	4.7×10^{-9}	4.6×10^{-8}
Lungs	2.0×10^{-10}	1.0×10^{-9}	1.0×10^{-8}
Muscle	6.0×10^{-9}	3.1×10^{-8}	3.0×10^{-7}
Ovaries	5.4×10^{-12}	2.8×10^{-11}	2.7×10^{-10}
Pancreas	2.9×10^{-11}	1.5×10^{-10}	1.5×10^{-9}
Prostate	3.6×10^{-12}	1.9×10^{-11}	1.8×10^{-10}
Red Bone Marrow	3.7×10^{-10}	1.9×10^{-9}	1.9×10^{-8}
Skin	3.8×10^{-10}	2.0×10^{-9}	1.9×10^{-8}
Spleen	4.3×10^{-11}	2.2×10^{-10}	2.1×10^{-9}
Testes	7.1×10^{-12}	3.7×10^{-11}	3.6×10^{-10}
Thymus	3.8×10^{-12}	2.0×10^{-11}	1.9×10^{-10}
Thyroid	3.8×10^{-12}	2.0×10^{-11}	1.9×10^{-10}
Uterus	2.8×10^{-11}	1.4×10^{-10}	1.4×10^{-9}
Remainder[a]	1.3×10^{-8}	6.8×10^{-8}	6.6×10^{-7}
Effective (ICRP)[b]	2.6×10^{-8}	1.4×10^{-7}	1.3×10^{-6}

[a]Based on the value for remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

[b]Calculated using the tissue weighting factors reported by ICRP (ICRP, 1990). The "effective" dose is defined in Section 11.0.

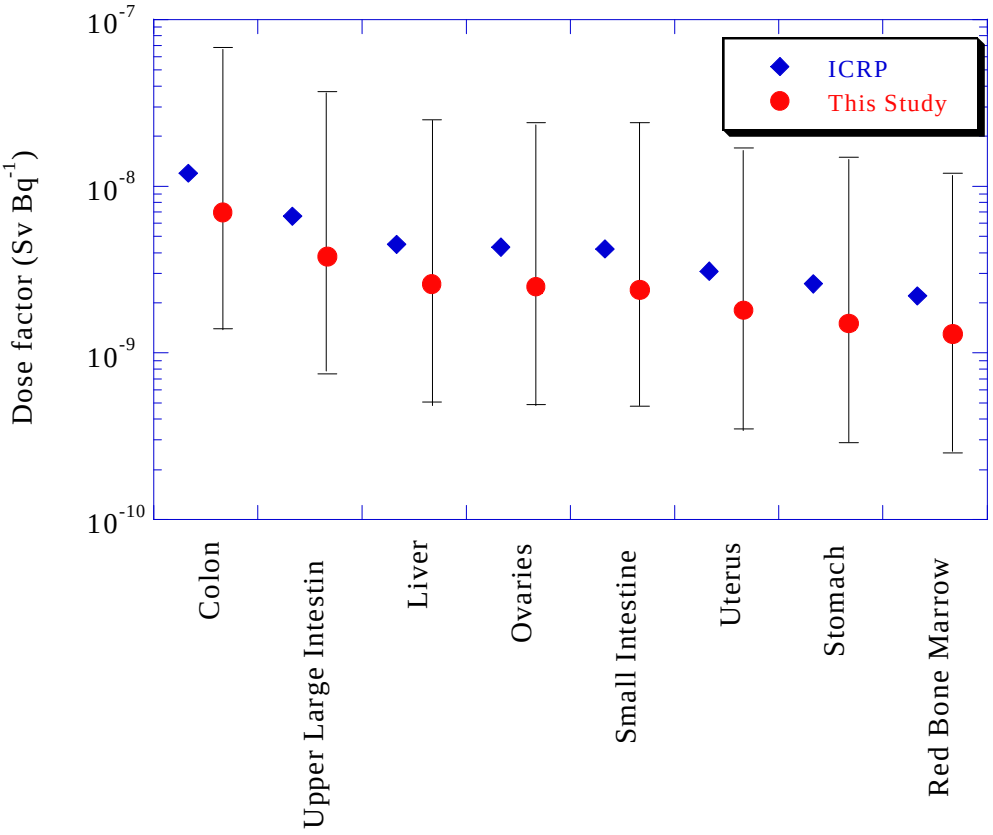


Figure 11.4.3 Comparison of the ingestion dose factors for ^{60}Co calculated in this study to the ICRP (1993) values (selected organs only)

11.4.5 Sensitivity Analysis

The sensitivity analysis indicated almost equal contributions to the total uncertainty in the dose factor for ^{60}Co: 54% for the biokinetics, and 46% for the fraction absorbed from the gastrointestinal tract to blood.

11.5 Ruthenium-106

Ruthenium-106 was determined during the initial screening analysis to be an important radioactive element released from the Oak Ridge Reservation via White Oak Creek. As a result, the contribution of ^{106}Ru to the overall dose possibly received by target individuals living along the Clinch River was required. The dose factor (DF) is one element of uncertainty in the dose. The purpose of this section is to assess the uncertainty in the DF, determine the most important factors contributing to the uncertainty in the DF for ^{106}Ru (sensitivity analysis), and describe the way in which this uncertainty will be treated in the risk assessment.

11.5.1 Background Information

Ruthenium is present in nature in low quantities. About 0.0004 ppm is present in the earth's crust (Merck, 1996). The most important isotopes obtained by the fission process of ^{235}U are ^{103}Ru and ^{106}Ru. Osmium is considered a chemical analog for ruthenium; however, neither of these elements influence human body metabolism. Normally, ruthenium is not present in the human body in measurable quantities.

11.5.2 Absorbed Fraction

Ruthenium is relatively uniformly distributed (not as uniformly as ^{137}Cs) in the human body, thus most of the organs are uniformly irradiated. The organs of the gastrointestinal tract will receive a larger dose because they are irradiated by both the ^{106}Ru distributed all over the body and by the ^{106}Ru passing through the gastrointestinal tract. As shown in Section 11.2.6 for a radionuclide that distributes fairly uniformly in the body, the contribution of the uncertainty in the absorbed fraction (*AF*) is small and can be neglected. Thus, as with ^{137}Cs, the uncertainty associated with the absorbed fraction (AF) of ^{106}Ru was neglected.

11.5.3 Biokinetics of ^{106}Ru in the Human Body

Most of the information available refers to experiments on mice, rats, rabbits, dogs and monkeys (Bruce and Carr, 1961; Bruce et al., 1962; Bruce, 1963; Furchner et al., 1971, Runkle et al., 1980), but some studies have been performed in man (Yamagata et al., 1969).

11.5.3.1 Absorption from the Gastrointestinal Tract

The physical and chemical state of ruthenium highly influences its absorption in the gastrointestinal tract. Experiments on rabbits and rats show that ruthenium bound in chemical complexes containing nitrogen, such as nitrosyl-ruthenium (RuNO) are absorbed to the greatest extent (average 8% - 13%, maximum 19%), while ruthenium in oxide or chloride forms is less readily absorbed (average 3%-5%, minimum 0.5%; Bruce and Carr, 1961; Bruce et al., 1962; Bruce, 1963). In addition, the same studies indicate a two to three times higher absorption rate if the animal is fasted. Other animal experiments indicated an absorption of 3 to 5% (Furchner et al., 1971). One study (Yamagata et al., 1969) measured the absorption of ruthenium in a male volunteer who ingested chloro-complexes of nitrosyl-ruthenium (III) and nitrosyl-ruthenium (IV) while consuming contaminated clams. The measured values in this case indicated an average absorption of 1%. Based on a review of the data, a range of 0.5% - 15% and a mode of 5% have been estimated.

11.5.3.2 Retention and Elimination of ^{106}Ru

Only a small fraction of the ingested ruthenium is absorbed, with the rest being rapidly eliminated by fecal excretion. Yamagata et al. (1969) reported that in one male volunteer studied, 95% of the administrated ruthenium was eliminated within 2 days.

Although the chemical form of ruthenium influences the degree of absorption, the distribution in the body is not greatly affected by the chemical form. The amount of ^{106}Ru absorbed by the gastrointestinal tract is small. However, once in the blood stream, a large fraction of ruthenium is eliminated in the urine. Bruce (1963) found in his experiments with rabbits and rats that about 50% of the absorbed ruthenium is excreted in the urine within 24 hours. The rest is stored in the body, where it remains with a long biological half-life. A short time (days) after absorption, ruthenium is more concentrated in the blood, the liver, and the kidneys than in other tissues, but the concentration becomes fairly uniformly distributed after 4-6 weeks (Furchner et al., 1971).

The total amount of ruthenium retained in the body decreases with time; a fast reduction of the inventory is observed within the first days after exposure, while a reduction at a much lower rate is observed at longer times after exposure. Experimentally, the retention function was described by a sum of exponential terms having different biological half-lives (Table 11.5.1, Figure 11.5.1). When ruthenium was administered orally, a sum of three exponentials was observed in the one subject studied (Yamagata et al., 1969), and a sum of either two or three exponential terms was observed in animal experiments (Furchner et al., 1971). Furchner et al. (1971) found a retention function as the sum of four exponential terms for ruthenium injected intravenously or intraperitoneally. The latter experiment is the most important, because it used ^{106}Ru and the followup was long enough (more than 400 days) that the long-term retention of ruthenium

Table 11.5.1 The whole-body retention of ruthenium from different experiments. The retention function is expressed as the sum of exponential terms, weighted by the coefficients a_i. The table lists the biological half-lives of each exponential term.

Species	Portal of Entry	Compound	Mechanism responsible for the Fastest Decay Term	Reference	a_1	τ [days]	a_2	τ [days]	a_3	τ [days]	a_4	τ [days]
Mice	I.P.	$^{106}RuCl_3$	urine excretion	A	0.36	0.3	0.39	5.2	0.1	33.2	0.15	815
Rats	I.P.	$^{106}RuCl_3$	urine excretion	A	0.16	0.3	0.37	5.7	0.28	22.1	0.19	517
Monkey	I.V.	$^{106}RuCl_3$	urine excretion	A	0.11	0.3	0.37	7.1	0.34	23.6	0.18	206
Dogs	I.V.	$^{106}RuCl_3$	urine excretion	A	0.18	0.3	0.4	11.1	0.24	53.7	0.19	1823
Mice	Oral	$^{106}RuCl_3$	fecal excretion	A	99.26	0.1	0.44	2.6	0.3	56.7	--	--
Rats	Oral	$^{106}RuCl_3$	fecal excretion	A	98.88	0.2	0.73	1.8	0.4	27.5	--	--
Monkey	Oral	$^{106}RuCl_3$	fecal excretion	A	99.14	0.3	0.85	15.2	--	--	--	--
Dogs	Oral	$^{106}RuCl_3$	fecal excretion	A	98.34	0.2	1.66	20.0	--	--	--	--
Man	Oral	^{103}Ru chloro-	fecal excretion	B	N/A	N/A	N/A	2.3	N/A	30	--	--
Man	blood	any	urine excretion	C	0.15	0.3	0.35	8.0	0.3	35.0	0.2	1000.0

[A] Furchner et al., 1971
[B] Yamagata et al., 1969
[C] ICRP, 1989

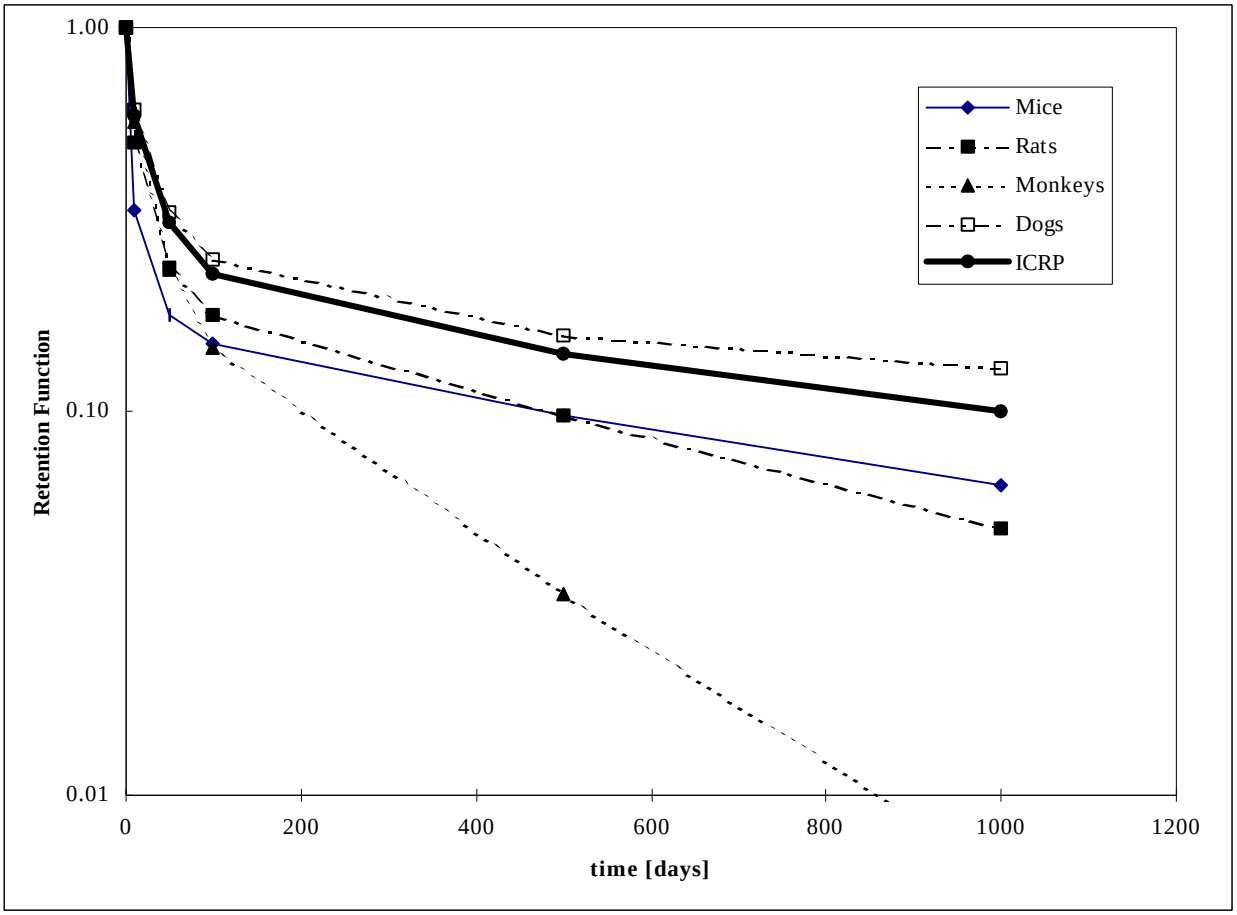


Figure 11.5.1 Comparison of the ruthenium retention functions from animal experiments (Furchner et al., 1971) and the retention function used by ICRP (1989).

could be determined. The long-term retention is the most important contributor to the total number of disintegrations, and therefore to the dose. A drawback of this experiment is that ruthenium was administered by injection. In this case, the retention function does not contain the effect of the passage through the gastrointestinal tract. This experiment is the one considered in the ICRP ruthenium model (Figure 11.5.2).

Direct usage of the Yamagata et al. (1969) human study in the assessment of the dose factor for ^{106}Ru is questionable for at least two reasons: (a) the study involved only one male subject, and (b) ^{103}Ru ($T_{1/2}$ = 40 days) was used in the study. As a result, the long-term retention of ruthenium in the body could not be observed.

The ICRP biokinetics model for ^{106}Ru (Figure 11.5.2) was used to estimate the uncertainties in the dose factors. The transfer rates between the compartments of the biokinetics model were centered on the ICRP recommended values, while the distributions (Table 11.5.2) developed for this report are based on the available data discussed above. Starting from the sets of animal data (Table 11.5.1), flow rates were determined for the biokinetic model (Figure 11.5.2). The minimum, maximum, and mean values of each flow rate were determined. An uncertainty factor for each flow rate was determined as the ratio between the maximum and mean value. These factors (Table 11.5.2) are less than 2 for all flow rates other than the one for the elimination rate from the long-term storage compartment, where a factor of 2.3 was found. In conclusion, a factor of 2 was used for all the flow rates, other than the elimination rate from the long-term storage compartment; that is, the ICRP nominal value was divided and multiplied by 2, loguniform distributions were assigned. For the long-term storage compartment, the minimum and maximum values were chosen as the limits of a loguniform distribution.

11.5.4 Results and Conclusions

In this work, the dose factors for ^{106}Ru were not recalculated, but only their uncertainties were estimated. To express the uncertainties in the ingestion dose conversion factors for ^{106}Ru, uncertainty factors were calculated and then applied to the current ICRP dose factors (Table 11.5.3).

Table 11.5.2 Probability distribution functions used to describe the uncertainty in the transfer rates in the $^{106}Ru/^{106}Rh$ biokinetic model (Figure 11.5.2). The gastrointestinal tract model is coupled with a metabolic model.

Transfer rates[a, b]	**Experimental uncertainty factor[c]**	**Nominal value ($days^{-1}$)**	**Lower Limit ($days^{-1}$)**	**Upper Limit ($days^{-1}$)**	**Distribution type**
Blood to A	1.03		0.4	1.62	loguniform
Blood to B	1.4		0.35	1.39	loguniform
Blood to C	1.1		0.23	0.92	loguniform
Blood to excretion	1.8		0.17	0.69	uniform
A to excretion	1.3		0.04	0.17	loguniform
B to excretion	1.3		0.01	0.04	loguniform
C to excretion	2.3		4.1×10^{-5}	3.5×10^{-3}	loguniform
Stomach to SI		24	12	57.6	triangular
SI to ULI and Blood		6	3.4	24	triangular
ULI to LLI		1.85	1.1	4.0	triangular
LLI to excretion		1	0.3	1.6	triangular
Absorption from gastrointestinal tract (f_1)		0.05	0.005	0.15	piece-wise uniform

[a]Both ^{106}Ru and ^{106}Rh are considered to follow the same biokinetic model.

[b]The abbreviations have the following meaning:
A = fast turnover body compartment
B = medium turnover body compartment
C = fast turnover body compartment
SI = small intestine
ULI = upper large intestine
LLI = lower large intestine

[c]This factor is the ratio between the maximum and the average observed value across all animal studies from Furchner et al. (1971).

Table 11.5.3 Organ dose delivered per unit of ingested activity of ^{106}Ru by adults (ingestion dose factors).

Target Organ	ICRP Reference Man[a] (Sv Bq^{-1})
Adrenals	**1.5×10^{-9}**
Bladder	**1.7×10^{-9}**
Bone surface	**1.5×10^{-9}**
Brain	**1.4×10^{-9}**
Breast	**1.4×10^{-9}**
Digestive System	
Oral Cavity	**1.5×10^{-9}**
Esophagus	**1.5×10^{-9}**
Stomach	**3.2×10^{-9}**
Small Intestine	**5.6×10^{-9}**
Upper Large Intestine	**2.6×10^{-8}**
Colon	**7.2×10^{-8}**
Rectum	**7.2×10^{-8}**
Gallbladder	**1.5×10^{-9}**
Kidneys	**1.5×10^{-9}**
Liver	**1.5×10^{-9}**
Lungs	**1.4×10^{-9}**
Muscle	**1.5×10^{-9}**
Ovaries	**1.7×10^{-9}**
Pancreas	**1.5×10^{-9}**
Prostate	**1.5×10^{-9}**
Red Bone Marrow	**1.5×10^{-9}**
Skin	**1.4×10^{-9}**
Spleen	**1.5×10^{-9}**
Testes	**1.5×10^{-9}**
Thymus	**1.4×10^{-9}**
Thyroid	**1.4×10^{-9}**
Uterus	**1.6×10^{-9}**
Remainder[b]	**1.5×10^{-9}**
Effective (ICRP)[c]	**7.0×10^{-9}**

[a]ICRP Publication 67 (ICRP,1993).

[b]Based on the value for remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

[c]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0.

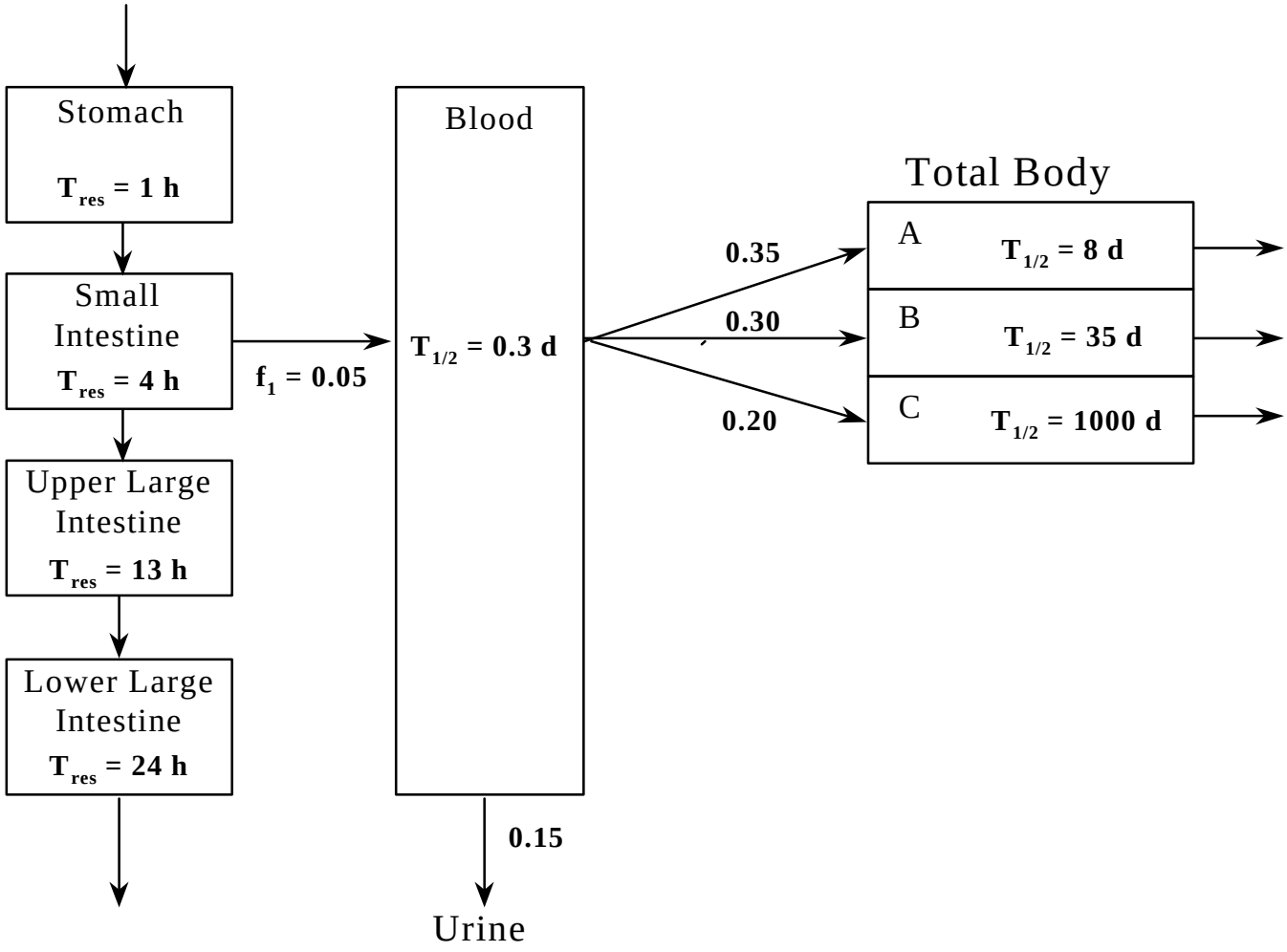


Figure 11.5.2 ICRP Metabolic model for ^{106}Ru (ICRP, 1989).

The uncertainty factors were estimated separately for the organs of the gastrointestinal tract and collectively for all other organs of the human body, in which ^{106}Ru is assumed to be distributed uniformly. The estimations are based on the number of nuclear disintegrations calculated for each component of the biokinetics model (Figure 11.5.2).

The dose to each organ of the human body other than gastrointestinal tract components is given by summing the energy deposited in that organ by ^{106}Ru and by its daughter ^{106}Rh from decays anywhere in the entire body. In this case, the source of radiation is the whole body, with each individual organ as a target. The source of uncertainty in the deposited energy is the number of nuclear disintegrations in the body.

On the other hand, the doses to the components of the gastrointestinal tract (GIT) were obtained by summing the energy deposited in these organs from the decay of ^{106}Ru/^{106}Rh in both the content of the gastrointestinal tract and in the rest of the body. Both the penetrating (gamma) and nonpenetrating (electrons) radiation emitted in the content of the gastrointestinal tract will contribute to the doses to the GIT component walls. Only the penetrating radiation generated in the rest of the body was assumed to reach the gastrointestinal tract components. However, the gastrointestinal tissues contain some of the systemic ^{106}Ru/^{106}Rh, which results in a small portion of penetrating radiation and nonpenetrating radiation irradiating these tissues (Leggett, 1997). In this study, the dose was obtained as a linear combination of the number of transformations in the gastrointestinal tract content and in the rest of the body. For the stomach, these contributors to the dose are comparable in magnitude. For the small intestine, upper large intestine, and colon the contribution of the ^{106}Ru/^{106}Rh in the content of these gastrointestinal tract components is dominant. This situation can be explained by noting that ruthenium has a low absorption from the gastrointestinal tract to the body fluid, and thus most of the ingested amount passes through the gastrointestinal tract. Also, this is apparent from the magnitude of the ICRP dose factors for these organs as compared to the dose factors for any other tissue (Table 11.5.3).

The Latin Hypercube Sampling technique using a sample size of 100 was used to propagate the errors and to produce a probability distribution function for the number of nuclear disintegrations for every organ of interest, including the content of the gastrointestinal tract. The biokinetics of ^{106}Ru was considered separately from the biokinetics of ^{106}Rh, but with the same transfer coefficients, which were sampled as fully correlated. The ratios of the 97.5th and 50th percentiles and of the 50th and 97.5th percentiles (Table 11.5.4) were used as estimates for uncertainty. These ratios show a tight range for the gastrointestinal tract (about a factor of 2), but a large uncertainty for the rest of the organs (a factor of 10 for soft tissues, and a factor of 13 for blood). For all organs other than the gastrointestinal components, the uncertainty should not exceed a factor of 10. This factor confidently covers the upper limit of the energy deposited (Table 11.5.4).

Table 11.5.4 Uncertainty factors in the number of nuclear disintegrations as a result of the uncertainties in the ICRP biokinetic model parameters.

	^{106}Ru/^{106}Rh	
	97.5%-tile/50%-tile	**50%-tile/2.5%-tile**
Stomach	1.8	1.8
Small Intestine content	2.1	2.1
Upper Large Intestine content	1.7	1.7
Lower Large Intestine content	2.1	1.5
Other organs	5.7	9.7

The values presented in Table 11.5.4 for the gastrointestinal tract are based only on the number of nuclear disintegrations in the contents of the tract. The ^{106}Ru/^{106}Rh dose factors for the small intestine, upper large intestine and colon are dominated by the number of disintegrations in their contents, but some contribution is also brought by the disintegrations in the walls of these organs and in the rest of the body. This is an additional source of uncertainty that must be taken into account. It is expected that the uncertainty in the dose factors for these organs should be about a factor of 3.

On the other hand, the dose to the stomach is given by comparable contributions of the energy emitted within the contents of the stomach (uncertainty of a factor of 2) and the energy emitted elsewhere in the body (uncertainty of a factor up to 10). Thus, an uncertainty factor for the stomach should have a value between 3 and 10. By assigning equal weights to the two contributors, this factor was estimated to have a value of about seven.

In conclusion, the uncertainties in the ^{106}Ru ingestion dose factors were estimated to a factor of 7 for stomach, a factor of 3 for intestines, and a factor of 10 for all other organs. Loguniform probability distributions were used to describe the uncertainty factors. The uncertainty factors were then applied to the current ICRP dose factors. The dose conversion factors are presented in Table 11.5.5 and Figure 11.5.3, and the energy per unit intake in Table 11.5.6.

Table 11.5.5 Estimated uncertainties in the organ-specific dose conversion factors for adults from ingestion of ^{106}Ru.

Target Organ	Present Study (Sv Bq^{-1})		
	95% Subjective Confidence Interval		
	lower bound	central value	upper bound
Adrenals	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Bladder	1.9×10^{-10}	1.7×10^{-9}	1.5×10^{-8}
Bone surface	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Brain	1.6×10^{-10}	1.4×10^{-9}	1.2×10^{-8}
Breast	1.6×10^{-10}	1.4×10^{-9}	1.2×10^{-8}
Digestive System			
Oral Cavity	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Esophagus	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Stomach	5.0×10^{-10}	3.2×10^{-9}	2.0×10^{-8}
Small Intestine	2.0×10^{-9}	5.6×10^{-9}	1.6×10^{-8}
Upper Large Intestine	9.2×10^{-9}	2.6×10^{-8}	7.4×10^{-8}
Colon	2.5×10^{-8}	7.2×10^{-8}	2.0×10^{-7}
Rectum	2.5×10^{-8}	7.2×10^{-8}	2.0×10^{-7}
Gallbladder	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Kidneys	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Liver	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Lungs	1.6×10^{-10}	1.4×10^{-9}	1.2×10^{-8}
Muscle	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Ovaries	1.9×10^{-10}	1.7×10^{-9}	1.5×10^{-8}
Pancreas	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Prostate	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Red Bone Marrow	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Skin	1.6×10^{-10}	1.4×10^{-9}	1.2×10^{-8}
Spleen	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Testes	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Thymus	1.6×10^{-10}	1.4×10^{-9}	1.2×10^{-8}
Thyroid	1.6×10^{-10}	1.4×10^{-9}	1.2×10^{-8}
Uterus	1.8×10^{-10}	1.6×10^{-9}	1.4×10^{-8}
Remainder[a]	1.7×10^{-10}	1.5×10^{-9}	1.3×10^{-8}
Effective (ICRP)[b]	7.8×10^{-10}	7.0×10^{-9}	6.2×10^{-8}

[a]Based on the value for remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

[b]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0 also on Table 11.5.6.

Table 11.5.6 Energy deposited in various organs from ingestion of 1 Bq of ^{106}Ru by adults.

Target Organ	Energy Deposited in Each Target from a Unit Intake [J/Bq]		
	95% Subjective Confidence Interval		
	lower bound	central value	upper bound
Adrenals	2.3×10^{-12}	2.1×10^{-11}	1.9×10^{-10}
Bladder	8.5×10^{-12}	7.6×10^{-11}	6.8×10^{-10}
Bone surface	2.0×10^{-11}	1.8×10^{-10}	1.6×10^{-9}
Brain	2.2×10^{-10}	2.0×10^{-9}	1.7×10^{-8}
Breast	5.6×10^{-11}	5.0×10^{-10}	4.4×10^{-9}
Digestive System			
Oral Cavity	1.7×10^{-11}	1.5×10^{-10}	1.3×10^{-9}
Esophagus	6.2×10^{-12}	5.5×10^{-11}	4.9×10^{-10}
Stomach	9.1×10^{-11}	5.9×10^{-10}	3.7×10^{-9}
Small intestine	1.5×10^{-9}	4.4×10^{-9}	1.2×10^{-8}
Upper Large Intestine	2.4×10^{-9}	6.7×10^{-9}	1.9×10^{-8}
Lower Large Intestine	4.0×10^{-9}	1.2×10^{-8}	3.3×10^{-8}
Rectum	8.9×10^{-10}	2.5×10^{-9}	7.1×10^{-9}
Gallbladder	1.7×10^{-12}	1.5×10^{-11}	1.3×10^{-10}
Kidneys	6.2×10^{-11}	5.6×10^{-10}	4.9×10^{-9}
Liver	3.6×10^{-10}	3.2×10^{-9}	2.9×10^{-8}
Lungs	1.9×10^{-10}	1.7×10^{-9}	1.5×10^{-8}
Muscle	5.3×10^{-9}	4.8×10^{-8}	4.2×10^{-7}
Ovaries	2.4×10^{-12}	2.2×10^{-11}	1.9×10^{-10}
Pancreas	2.0×10^{-11}	1.8×10^{-10}	1.6×10^{-9}
Prostate	2.7×10^{-12}	2.4×10^{-11}	2.1×10^{-10}
Red Bone Marrow	3.0×10^{-10}	2.7×10^{-9}	2.4×10^{-8}
Skin	4.9×10^{-10}	4.4×10^{-9}	3.9×10^{-8}
Spleen	3.6×10^{-11}	3.2×10^{-10}	2.9×10^{-9}
Testes	6.6×10^{-12}	5.9×10^{-11}	5.3×10^{-10}
Thymus	3.8×10^{-12}	3.4×10^{-11}	3.0×10^{-10}
Thyroid	3.8×10^{-12}	3.4×10^{-11}	3.0×10^{-10}
Uterus	1.7×10^{-11}	1.5×10^{-10}	1.3×10^{-9}
Remainder[a]	9.8×10^{-9}	8.8×10^{-8}	7.7×10^{-7}
Effective (ICRP)[b]	5.4×10^{-8}	4.8×10^{-7}	4.3×10^{-6}

[a]Based on the value for remainder as reported by ICRP Publication 67 (ICRP,1993). The "remainder" is defined in Section 11.0.

[b]Calculated using the tissue weighting factors reported by ICRP (ICRP,1990). The "effective" dose is defined in Section 11.0 also on Table 11.5.6.

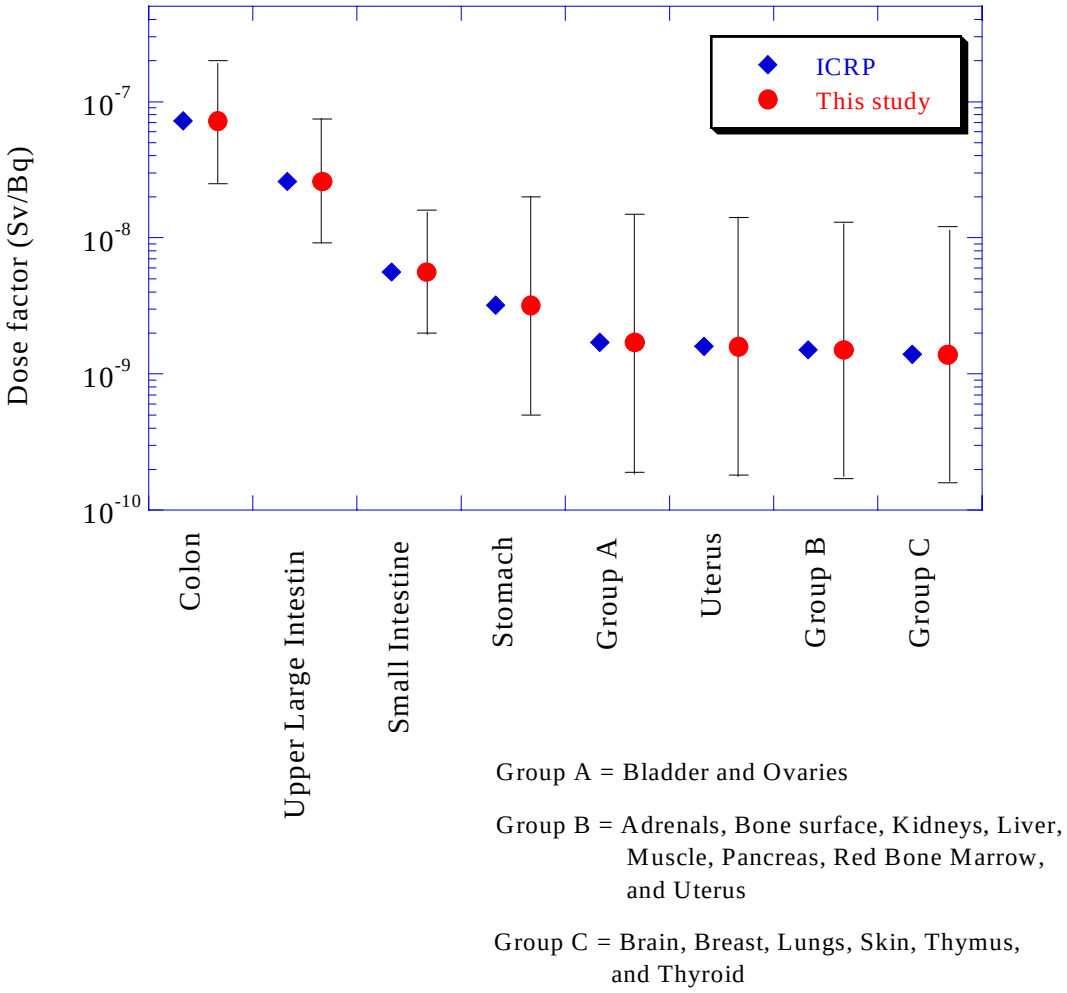


Figure 11.5.3 Comparison of the ingestion dose factors for ^{106}Ru calculated in this study to the ICRP (1993) values

11.5.5 Sensitivity Analysis

Since absorbed ruthenium is distributed fairly uniformly in the body, a sensitivity analysis was performed for the "body tissue" as a description of all organs. The organs of the digestive system were considered separately, as indicated by the metabolic model for ruthenium (Figure 11.5.4). The sensitivity analysis is based on the number of nuclear transformations as a function of the biokinetic parameters (Table 11.5.7).

For every component of the digestive system, the rate at which the radionuclide is cleared from that compartment is the dominant source of uncertainty in the organ dose. Other important contributors are the absorption fraction for the small intestine, and elimination rates from stomach and small intestine for colon (Table 11.5.7).

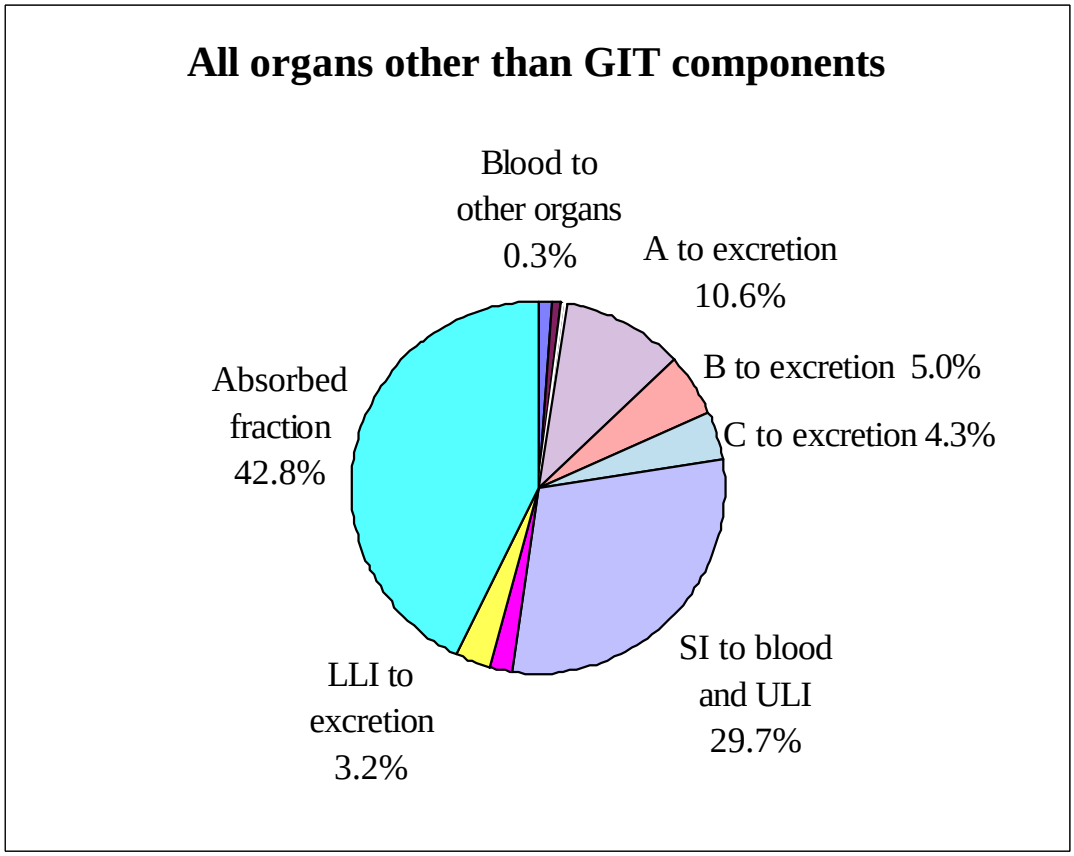
For all soft tissues (Figure 11.5.4) the most important contributor to uncertainty in the organ doses is the absorbed fraction from the gastrointestinal tract to blood (f_1), followed by the elimination rate from the small intestine and by the elimination rates from the body tissues.

Table 11.5.7 Percent contribution of the various biokinetic parameters to the uncertainty in the number of nuclear transformations of ^{106}Ru in each specified organ.

Parameter[a]	Other Organs	Stomach content	Small Intestine content	Upper Large Intestine content	Lower Large Intestine content
Blood to A	1.1	--[b]	--	--	--
Blood to B	0.8	--	--	--	--
Blood to C	0.4	--	--	--	--
Blood to excretion	0.3	--	--	--	--
A to excretion	**10.6**	--	--	--	--
B to excretion	5.0	--	--	--	--
C to excretion	4.3	--	--	--	--
Stomach to Small Intestine	0.0	**100.0**	0.4	1.2	6.4
Small Intestine to blood and Upper Large Intestine	**29.7**	--	**92.9**	0.2	**10.7**
Upper Large Intestine to Colon	1.8	--	--	**98.3**	0.5
Colon to excretion	3.2	--	--	--	**82.2**
Absorbed fraction	**42.8**	--	6.7	0.4	0.2

[a]A = fast turnover body compartment; B = Medium turnover body compartment; C = Slow turnover body compartment
[b]not applicable

LEGEND
A = Fast turnover body compartment
B = Medium turnover body compartment
C = Slow turnover body compartment
SI = Small intestine
ULI = Upper Large Intestine
LLI = Lower Large Intestine (Colon)



LEGEND

A = Fast turnover body compartment
B = Medium turnover body compartment
C = Slow turnover body compartment
SI = Small Intestine
ULI = Upper Large Intestine
LLI = Lower Large Intestine

Figure 11.5.4 Contributors to the uncertainty in the number of nuclear transformations for all organs other than the gastrointestinal (GIT) components.

11.6 Iodine-131

Iodine-131 was determined during the initial screening analysis to be an important radioactive element released from the Oak Ridge Reservation via White Oak Creek. As a result, the dose contribution of ^{131}I to the overall dose possibly received by target individuals living along the Clinch River was required. The doses to the thyroid per unit intake of ^{131}I (dose factors) have been calculated for various situations (Dunning and Schwartz, 1981; Killough and Eckerman, 1986; ICRP, 1990). The calculations have been reproduced in this work in order to identify the most important contributors to the uncertainty in the dose factor (DF) and to attempt to make use of the most recent data on thyroid mass obtained by ultrasonography methods.

11.6.1 Background Information

Iodine is an important chemical element in human metabolism. The main source of stable iodine for humans is iodized salt, but natural iodine is available in food items (milk, agricultural produce) and in drinking water.

11.6.2 Absorbed Fraction

The largest average energy from an electron emission for ^{131}I decay is approximately 0.35 MeV and occurs less than 0.2% of the time (see Appendix 10-A for more details). The distance traveled by this electron is about 1 mm (Turner, 1986). Iodine is not uniformly distributed in the thyroid itself, primarily accumulating in the follicles of the thyroid. The follicles are surrounded by follicular cells, connective tissue, and blood vessels and are located further than 1 mm deep inside the thyroid gland. As a consequence, the electrons emitted by each nuclear transformation of the ^{131}I are assumed to be completely absorbed by the thyroid gland.

Part of the energy of the penetrating radiation (gamma and x-rays) escapes from the thyroid. The amount of energy deposited in the thyroid gland and its dependence on the size of the gland are described by the absorbed energy fraction, which is described in Section 11.6.3.1.

11.6.3 Biokinetics of ^{131}I in the Human Body

Ingested iodine is completely absorbed from the gastrointestinal tract into the bloodstream, where ^{131}I is considered to be metabolized similarly to stable iodine. Usually less than one-fourth of the iodine in the blood stream is cleared by the thyroid gland, while about three-fourths is collected by the kidneys and excreted in the urine.

The thyroid gland is regulated by the thyroid-stimulating hormone (TSH) produced by the anterior pituitary gland. The passage of the iodine from the extracellular fluid (supplied by blood) into the thyroid cells and follicles is controlled by the basal membrane of the thyroid cell, which traps the iodide. The concentration of iodine in the cells of a thyroid functioning at normal levels is about 30 times larger than the concentration in blood. The thyroid gland uses iodine for secretion of two significant hormones, thyroxine (T4) and

triiodothyronine (T3), which have very important roles in controlling the metabolic rate of the body. Complete lack of thyroid secretion usually causes the basal metabolic rate to fall to about 40% below normal, and extreme excess of thyroid hormones can cause the basal metabolic rate to rise as high as 60 to 100% above normal (Guyton, 1991).

After formation, the major thyroid hormones are stored inside the gland but outside the producing cells, in a follicular colloid, in an amount sufficient to supply the body with its normal requirements of thyroid hormones for 2 or 3 months (Guyton, 1991). ICRP (1979) recommends use of a value of 80 days for the half-life of iodine in the thyroid compartment, to account for the storage period of hormones in the thyroid. This long storage time allows most of the ^{131}I to decay, thus irradiating the thyroid gland.

Once the hormones are released to the blood, they are transported to cells all over the body, where they are metabolized. Metabolization of the hormones releases the iodine, which again enters the bloodstream. However, due to the long storage time of the hormones in the thyroid gland as compared to the short half-life of ^{131}I, very little ^{131}I is actually recycled in this manner, so that recycling contributes little to the radioiodine dose.

Certain characteristics of the thyroid gland, such as mass, are dependent on the age of the individual. The lower mass of the thyroid during childhood makes the dose to a child's thyroid appreciably higher than the dose to an adult's thyroid for the same amount of iodine ingested. Therefore, children are the critical group of the population.

11.6.3.1 Modeling Approach

The dose to the thyroid can be calculated using ICRP methodology (Eckerman, 1994; Killough and Eckerman, 1986). The activity burden in the thyroid depends on the age and sex of the individual. The activity calculations are based on a two-compartment biokinetic model (Figure 11.6.1).

The dose factor is given by the following equation:

$$DF = IA \cdot SEE \qquad \text{(Eq. 11.6.1)}$$

$$IA = \frac{(a \cdot \lambda_1)}{(\lambda_R - \lambda_1)(\lambda_R - \lambda_2)} \qquad \text{(Eq. 11.6.2)}$$

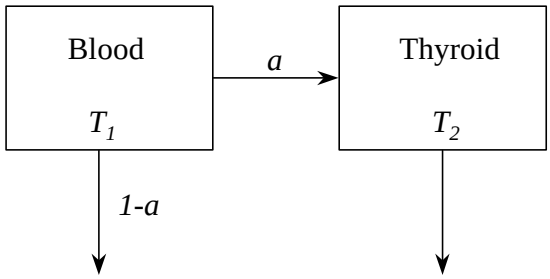


Figure 11.6.1 Two-compartment metabolic model for iodine (Killough and Eckerman, 1986). T_1 and T_2 are the biological removal half-lives for each compartment, and *a* is the fraction of material transferred to the thyroid.

where

DF	=	the dose factor (Gy Bq^{-1})*;
IA	=	the integrated activity burden of ^{131}I in the thyroid for a unit intake (d);
SEE	=	specific effective energy (Sv Bq^{-1} d^{-1})*;
a	=	the fractional uptake of ^{131}I from the blood to the thyroid (unitless);
λ_1	=	$\ln(2)/T_1$ ($days^{-1}$), T_1 = biological half-life of ^{131}I in the blood (d);
λ_2	=	$\ln(2)/T_2$ ($days^{-1}$), T_2 = biological half-life of ^{131}I in the thyroid gland (d); and
λ_R	=	$\ln(2)/T_{1/2}$ ($days^{-1}$), $T_{1/2}$ = radioactive half-life of ^{131}I (d).

The specific effective energy represents the dose rate per unit activity in the thyroid tissue, and it depends on the fraction of energy absorbed in the gland (Equation 11.6.2). Estimates for the *SEE* are reported by Killough and Eckerman (1986). The largest contribution to the *SEE* is given by the nonpenetrating radiation (â radiation, conversion electrons, and Auger electrons), which is assumed to be completely absorbed in the thyroid. The contribution of the penetrating radiation (gamma and x-radiation) is much less important. The *SEE* is a function of thyroid mass (M_{th}) and was represented by this equation:

$$SEE = \frac{g}{M_{th}} + h \cdot (M_{th})^{p} \qquad \text{(Eq. 11.6.3)}$$

The fitted parameters used in this analysis were $g = 0.406$, $h = 0.01014$, and $p = -0.666$ (Killough and Eckerman, 1986), which provided the expression of M_{th} in grams and the *SEE* in rads mCi^{-1} h^{-1}.

The half-lives of iodine in the blood (T_1) and thyroid (T_2) are correlated parameters. The correlation is given by the following set of equations:

$$\lambda_1 = \frac{s}{1 - r} - \lambda_R \qquad \text{(Eq. 11.6.4a)}$$

and

$$\lambda_2 = \frac{(s \cdot r)}{1 - r} - \lambda_R \qquad \text{(Eq. 11.6.4b)}$$

*Iodine-131 emits beta and gamma radiation only. No alpha particles or neutrons are emitted. In this case, the expressed dose expressed in Gy is equivalent to the dose expressed in Sv.

where

$$s = -\frac{\ln(r)}{t_{max}} \qquad \text{(Eq. 11.6.5)}$$

and "r" is defined by the equation

$$\frac{\left[1 - r^{\frac{t_2}{t_{max}}}\right] \cdot r^{\left(\frac{r}{1-r} \cdot \frac{t_2}{t_{max}}\right)}}{\left[1 - r^{\frac{t_1}{t_{max}}}\right] \cdot r^{\left(\frac{r}{1-r} \cdot \frac{t_1}{t_{max}}\right)}} = exp\left[-\frac{0.693}{T^*} \cdot (t_2 - t_1)\right] \qquad \text{(Eq. 11.6.6)}$$

The index "1" represents the blood compartment, and the index "2" represents the thyroid compartment. The equations were derived by Killough and Eckerman (1986) on the basis of measurements of an "apparent" biological half-life of ^{131}I in the human body (T^*) as a function of age and sex. The parameter $t_{max} = t_1 = 1$ day represents the constraint that the activity in the thyroid has its maximum 1 day after the intake. Killough and Eckerman (1986) showed that a variation in this parameter from 0.5 to 2 days produces a deviation in the results of less than 11%. The parameters $t_1 = 1$ day and $t_2 = 15$ days define the interval in which the biological half-life is usually measured.

11.6.3.2 Biokinetic Model Parameters

A number of input parameters must be determined to completely solve the system of equations (Eq. 11.6.1 ÷ Eq. 11.6.6). The first is the measured "apparent" biological half-life (T^*), and the second is the fraction of iodine taken up by the thyroid from the blood (a). The values for (a) and (T^*) used in this work (Table 11.6.1) are based on those recommended by Dunning and Schwartz (1981). To avoid values of the uptake fraction (a) larger than 1, a triangular distribution has been used. The data were interpolated between age groups using a piecewise linear interpolation technique.

Table 11.6.1 Distribution assigned to the biological half-life and the fractional uptake for the uncertainty analysis.

Parameter		Units	Distribution			Shape
			Lower	Median	Upper	
Newborn	T^*	d	4.0[a]	13.0	44.0[a]	Lognormal
	a	unitless	0.17	0.37	0.81	Triangular
Child	T^*	d	3.0[a]	10.0	42.0[a]	Lognormal
	a	unitless	0.17	0.37	0.81	Triangular
Adolescent	T^*	d	16.0[a]	44.0	119.0[a]	Lognormal
	a	unitless	0.2	0.43	0.94	Triangular
Adult	T^*	d	24.0[a]	72.0	220.0[a]	Lognormal
	a	unitless	0.08	0.17	0.37	Triangular

[a]"Lower" and "upper" represent the 2.5 percentile and the 97.5 percentile of the lognormal distribution, respectively.

The third parameter is the mass of the thyroid as a function of age. Numerous studies report measurements of the thyroid mass obtained by autopsy (Mochizuki et al., 1963; Kereiakes et al., 1965; Kay et al., 1966; Dunning and Schwartz, 1981; Killough and Eckerman, 1986). However, most of the sets of measurements are incomplete because either some age groups are not covered, or few measurements are available for a specific age group.

Modern ultrasonography methods are currently used to determine the volume of the thyroid in large size populations. The mass can be calculated from the measured volume using the density of the thyroid tissue. Thus, better distributions in the population are expected to become available. Moreover, in some cases, the ultrasonography method has shown smaller thyroid masses than those usually obtained by autopsy (Gutekunst et al., 1986; Likhtarev et al., 1993). Such a result will produce larger dose estimates. Some ultrasonographic measurements performed in the Former Soviet Union in areas affected by deposition from the Chernobyl accident were investigated. These measurements do not conclusively indicate a potential bias of the thyroid mass towards lower values (Yureiva et al., 1994; Derzhitskiy et al., 1994; Averichev et al., 1994; Avramenko et al., 1994; Danilyuk, 1994). A more recent review of the available ultrasound data (Table 11.6.2) was performed by Bier (1996). He showed a clear trend towards lower values of the thyroid mass obtained by modern ultrasonography methods as compared to the corresponding data obtained from autopsy (Figure 11.6.2).

Table 11.6.2 Thyroid mass (grams) obtained by ultrasound techniques (Bier, 1996).

Age in Years	Number of Measurements	Mean Thyroid Mass	Standard Deviation
Newborn	33	1.89	0.40
1	30	2.05	0.60
2	12	2.10	0.50
3	16	2.50	0.70
4	65	2.70	0.20
5	18	3.40	1.00
6	56	3.10	1.25
7	223	4.70	0.21
8	400	5.60	0.42
9	266	5.70	0.17
10	289	6.90	0.60
11	236	7.80	0.69
12	261	8.80	0.24
13	6740	9.10	1.62
14	209	11.6	0.89
15	415	11.5	0.29

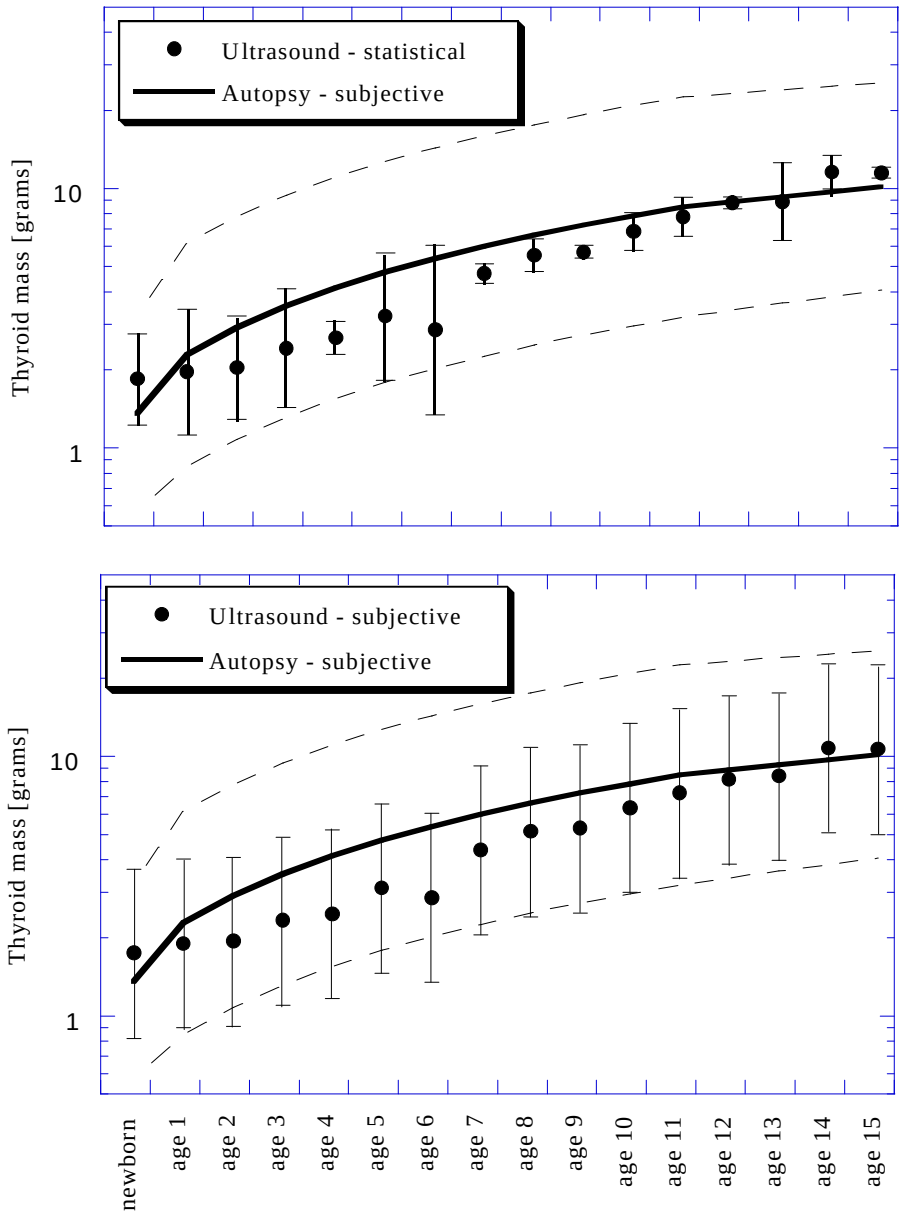


Figure 11.6.2 Comparison of the thyroid mass obtained by autopsy investigation (Killough and Eckerman, 1996) and by ultrasound investigations (Bier, 1996). The 95% confidence intervals are included. For autopsy data, the dashed lines represent the 95% subjective confidence interval. For ultrasound data, the vertical lines represent the 95% confidence interval from statistical analysis of data, and the dashed lines represent the 95% subjective confidence interval obtained from consideration of the inter-individual uncertainty.

The values reported by Bier (1996) are weighted values across several reviewed studies (weight based on the numbers used in each study). For a child of age 6, the thyroid mass from Bier (1996) is unrealistically lower in comparison to the values in the same study for children ages 5 and 7. Therefore, the observed thyroid mass for age 6 was replaced by the average between values for ages 5 and 7.

However, Bier (1996) did not address the issue of interindividual variability of the thyroid mass. To investigate this issue, the data from Bier (1996) were compared to data from the countries of the Former Soviet union (FSU; Yureiva et al., 1994; Derzhitsky et al., 1994; Averichev et al., 1994; Avramenko et al., 1994; Danilyuk, 1994; Figures 11.6.3, 11.6.4, 11.6.5). The latter data report the distributions of the thyroid mass in the population of children from areas affected by the Chernobyl accident. These distributions show a trend toward higher values of thyroid mass, but their interindividual variability should be representative for any population of children. A geometric standard deviation (GSD) was obtained from the FSU data. A GSD of 1.5 was determined to be the best representation of interindividual variability of the thyroid mass (Figures 11.6.3, 11.6.4, 11.6.5).

In conclusion, the uncertainty in the thyroid mass is expressed using lognormal distributions having average values from Bier (1996; Table 11.6.2) and geometric standard deviations of 1.5 applied to each age group.

11.6.4 Results and Conclusions

The estimates of the 95% subjective confidence intervals for the age-specific dose conversion factors are listed in Table 11.6.3 and plotted in Figure 11.6.6. The results are in good agreement with similar results reported by Snyder at al. (1994) and by ICRP (1993).

11.6.5 Sensitivity Analysis

The results of the sensitivity analysis indicating the major contributors to the uncertainty in the estimates of the dose factors are shown in Figure 11.6.7. The analysis was performed only for selected age groups, given the relatively low variation of the parameters as the age increases. For all ages, the major contributor to the uncertainty is the thyroid mass, followed by the fractional uptake (a). Due to the relationship between the biological half-lives, both are shown as having equal contributions. Their contribution becomes lower as the age increases.

Table 11.6.3 Age-specific dose factors (Sv Bq^{-1}) for ingestion of ^{131}I determined in this study.

	95% Subjective Confidence Interval		
Age in Years	**lower bound**	**central value[a]**	**upper bound**
Newborn	1.0×10^{-6}	3.0×10^{-6}	1.1×10^{-5}
1	7.1×10^{-7}	2.4×10^{-6}	7.2×10^{-6}
2	8.3×10^{-7}	2.5×10^{-6}	7.1×10^{-6}
3	6.1×10^{-7}	2.1×10^{-6}	7.3×10^{-6}
4	7.0×10^{-7}	2.2×10^{-6}	5.4×10^{-6}
5	6.1×10^{-7}	1.8×10^{-6}	5.2×10^{-6}
6	4.8×10^{-7}	1.6×10^{-6}	5.0×10^{-6}
7	5.1×10^{-7}	1.5×10^{-6}	3.9×10^{-6}
8	4.5×10^{-7}	1.4×10^{-6}	3.7×10^{-6}
9	5.3×10^{-7}	1.4×10^{-6}	3.9×10^{-6}
10	4.8×10^{-7}	1.3×10^{-6}	3.6×10^{-6}
11	4.7×10^{-7}	1.3×10^{-6}	3.6×10^{-6}
12	3.7×10^{-7}	1.1×10^{-6}	3.1×10^{-6}
13	3.5×10^{-7}	9.8×10^{-7}	2.7×10^{-6}
14	2.7×10^{-7}	7.3×10^{-7}	2.0×10^{-6}
15	2.4×10^{-7}	7.0×10^{-7}	1.8×10^{-6}

[a] median

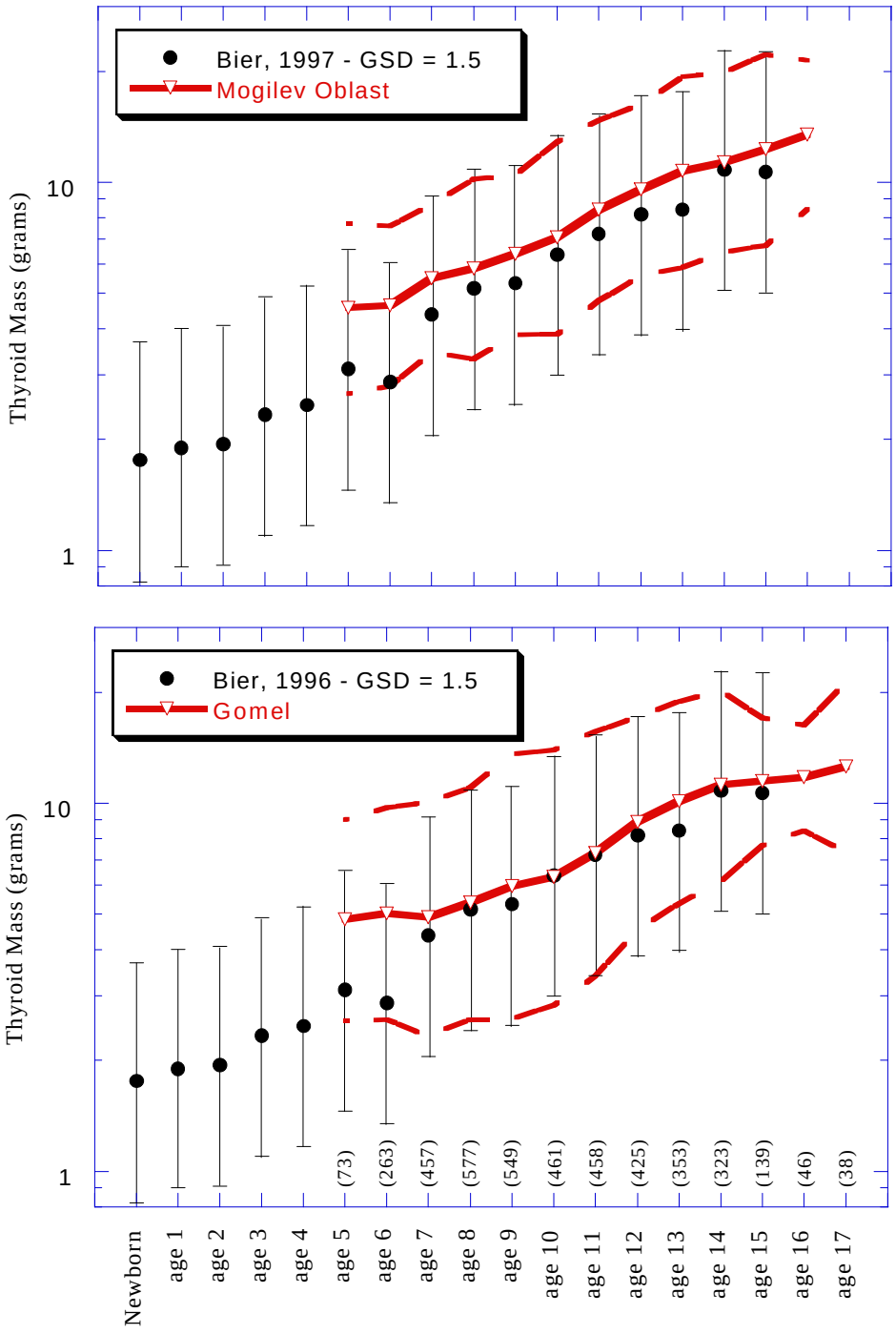


Figure 11.6.3 Comparison of thyroid masses from Bier (1996) and data from Mogilev and Gomel Counties (oblast) from the Former Soviet Union affected by the Chernobyl accident. Lognormal distributions with geometric standard deviations of 1.5 were assigned to the average thyroid masses from Bier (1996). The upper and lower bounds delimit a 95% confidence interval.

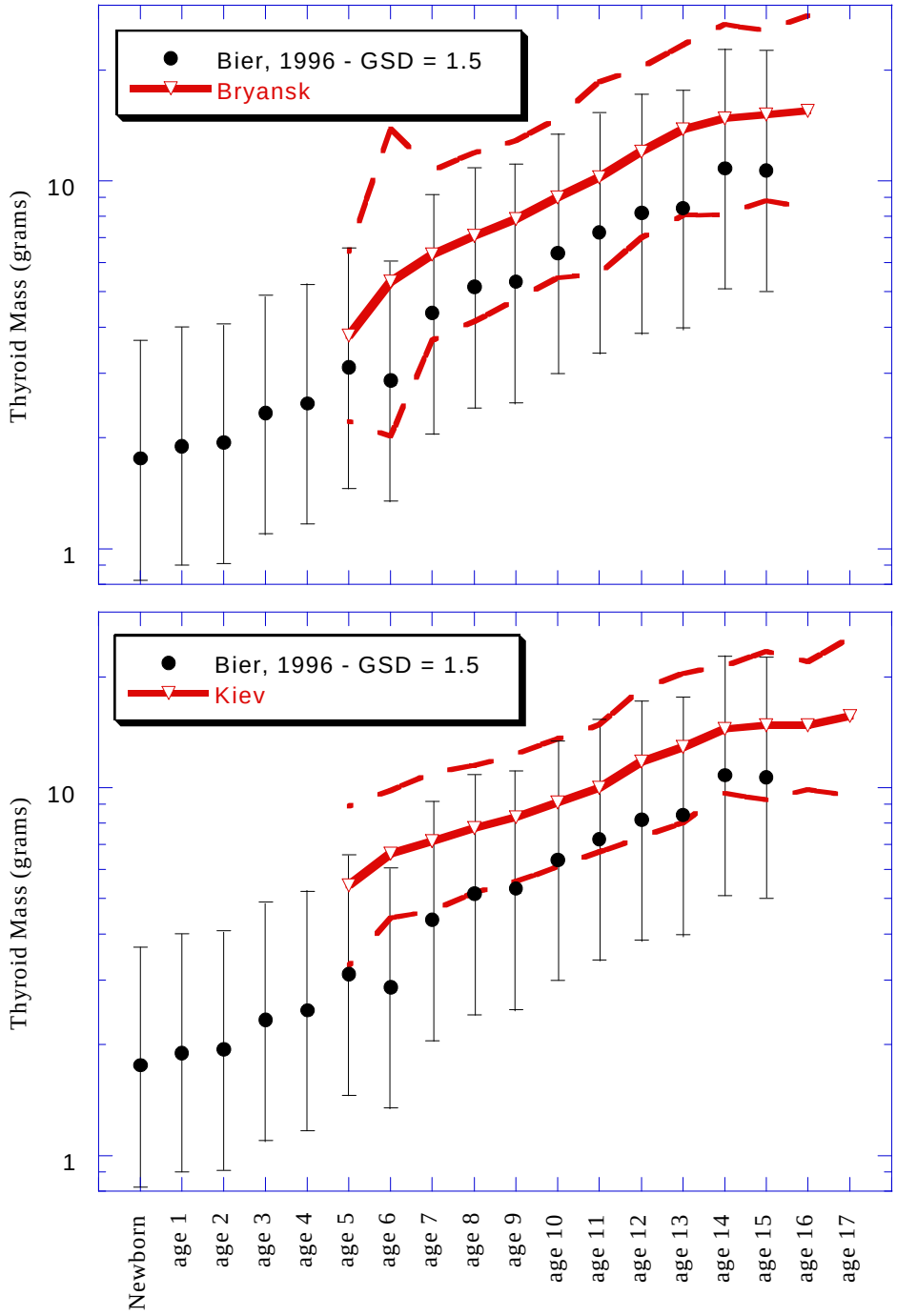


Figure 11.6.4 Comparison of thyroid masses from Bier (1996) and data from Bryansk and Kiev Counties (oblast) from the former Soviet Union affected by the Chernobyl accident. Lognormal distributions with geometric standard deviations of 1.5 were assigned to the average thryoid masses from Bier (1996). The upper and lower bounds delimit a 95% confidence interval.

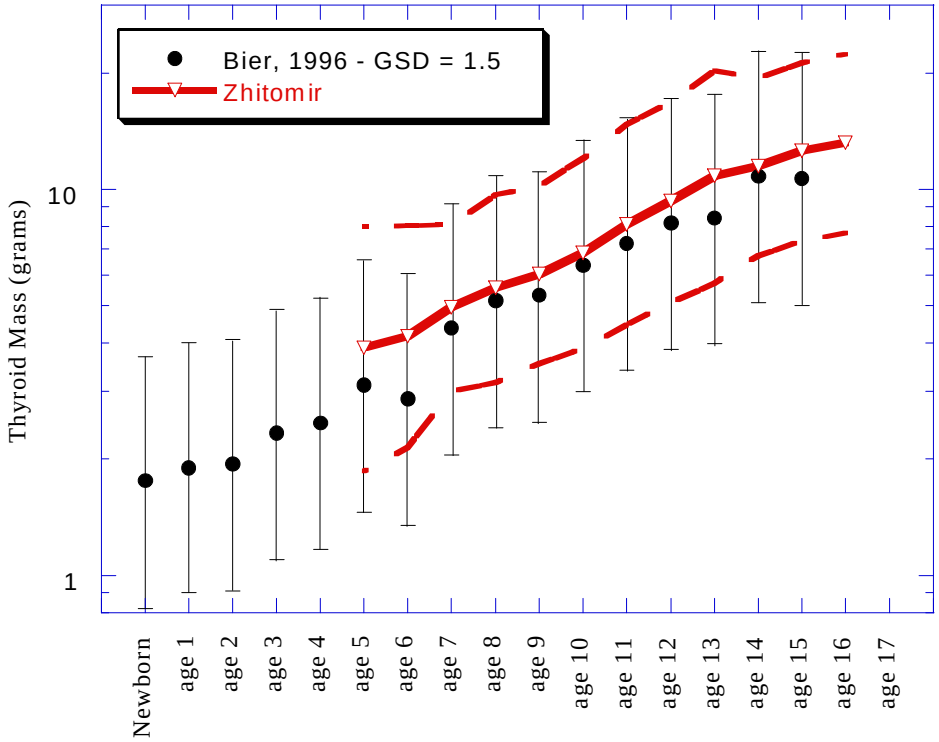


Figure 11.6.5 Comparison of thyroid masses from Bier (1996) and data from Zhitomir County (oblast) from the Former Soviet Union affected by the Chernobyl accident. Lognormal distributions with geometric standard deviations of 1.5 were assigned to the average thyroid masses from Bier (1996). The upper and lower bounds delimit a 95% confidence interval.

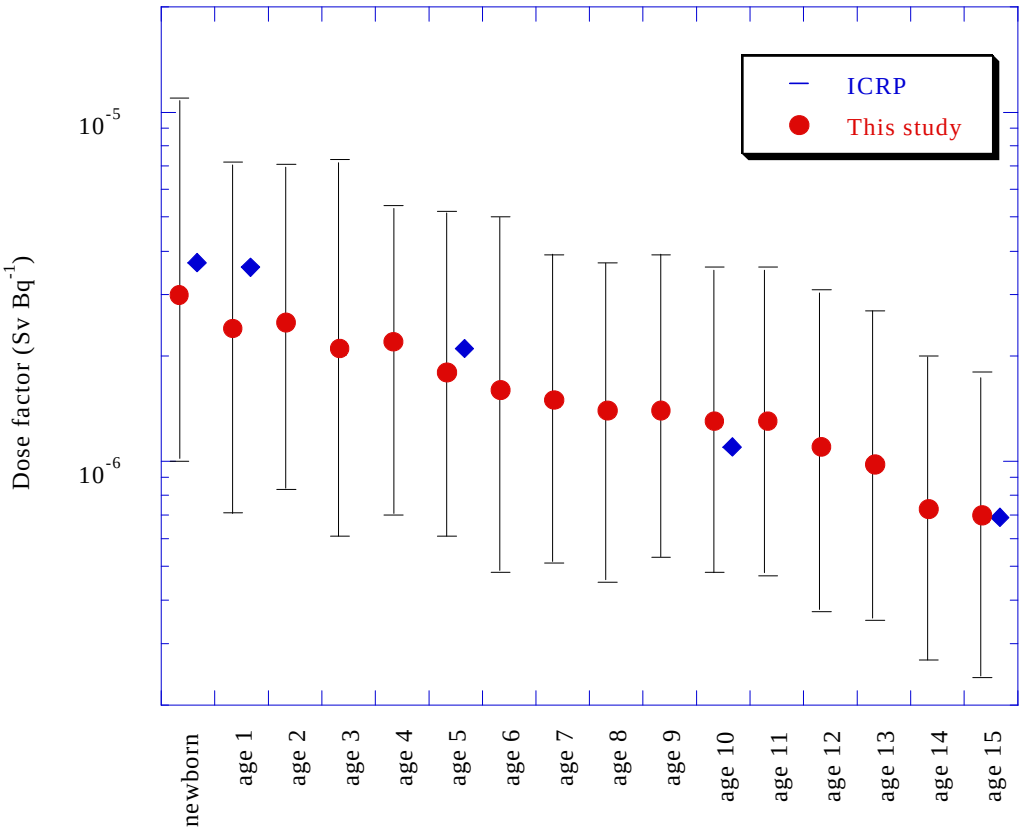


Figure 11.6.6 Comparison of the ingestion dose factors for ^{131}I calculated in this study to the ICRP (1993) values (doses to thyroid only)

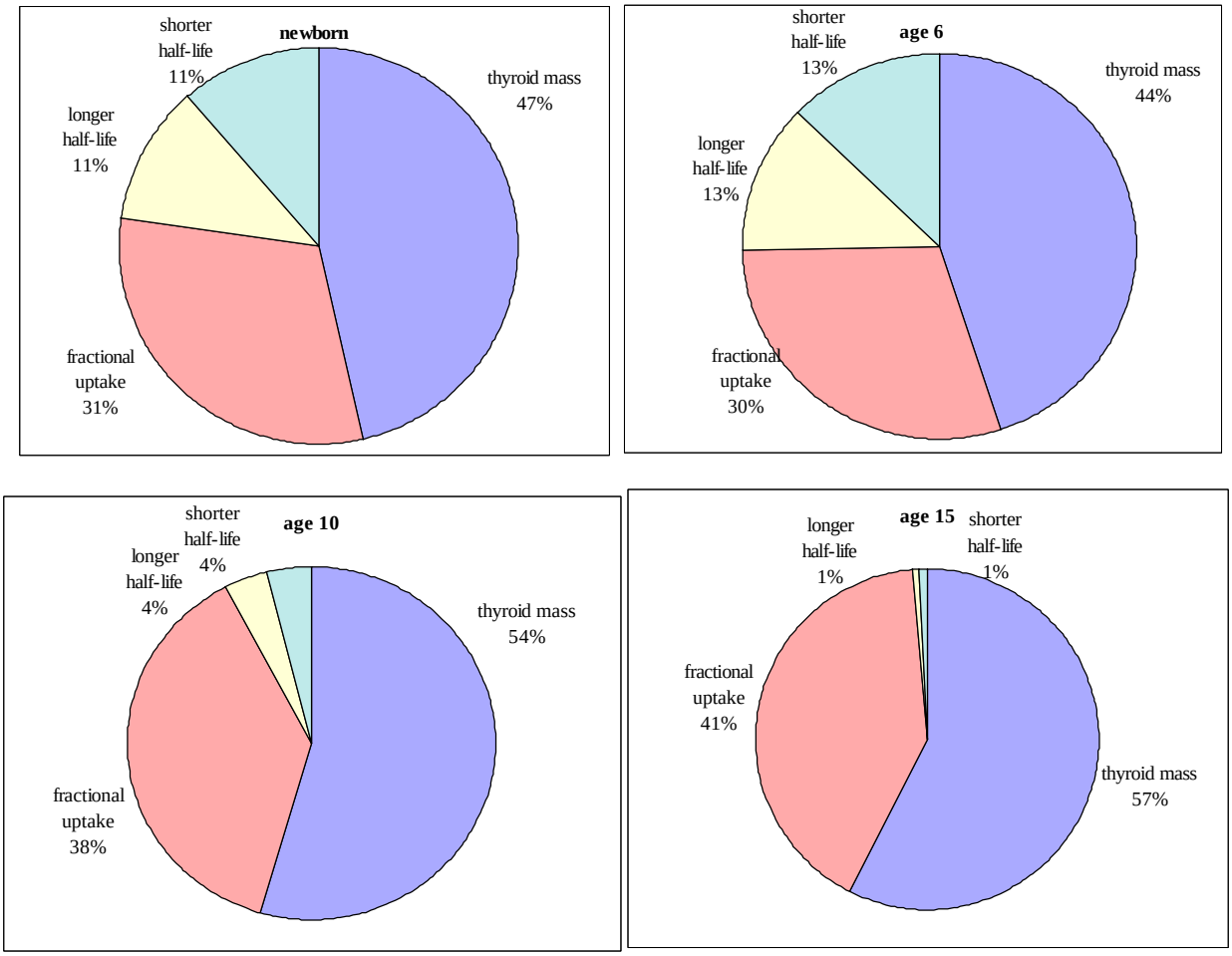


Figure 11.6.7 Contributors to the uncertainty in the ^{131}I ingestion dose factor for various age groups.

11.7 Summary

This section describes the scientific basis and the methodology used for the derivation of the radiation doses to various organs from ingestion of a unit of activity of a given radionuclide. The doses per unit intake are referred to as "dose factors." The purpose of this section is to develop *confidence intervals* for the ingestion dose factors for selected radionuclides, and also to identify the dominant sources of uncertainty.

The target individuals considered are adults for ^{137}Cs, ^{90}Sr, ^{60}Co and ^{106}Ru and children ages 0-15 for ^{131}I. The dose factors were estimated for twenty-two organs for ^{137}Cs, ^{90}Sr, and ^{60}Co and for the thyroid in the case of exposure to ^{131}I.

Different approaches were employed to determine the dose factors, according to the amount and quality of the information available for each radionuclide. ICRP biokinetic models (ICRP, 1993) were used for ^{90}Sr and ^{106}Ru. For ^{90}Sr, new dose factors were calculated, while, for ^{106}Ru, the ICRP model was used only to determine the uncertainty introduced by the biokinetics of this isotope in the body. The uncertainty in biokinetics was then applied as an uncertainty factor to the ingestion dose factors published by ICRP (1993) for ^{106}Ru.

For ^{137}Cs and ^{60}Co, the simple functional forms of the retention of the isotopes in the human body were used to determine the uncertainty in the dose factors. The relationship between the retention time of ^{137}Cs in the body and the amount of potassium in the body was explicitly considered, in order to reduce the uncertainty. For both of these isotopes, uncertainty factors were first derived and then applied to the published ICRP dose factors.

To properly account for the case of exposure to multiple radionuclides, the uncertainty induced by the variability in the mass of the organs is not included in the ingestion dose factors reported in this section. However, the uncertainty in the organ masses is included separately in the final results using the technique described in Section 11.1.

A simple model, formed by two-compartments (plasma and thyroid), was used for ^{131}I. New thyroid dose factors were estimated in this study by taking into account the relationship between various physiological parameters and using the most recent measurements of thyroid mass obtained by ultrasound measurements.

Uncertainties in the model parameters are introduced both by the inter-individual variability in the population, and by the lack of complete knowledge about the physical and physiological parameters governing the kinetics of the radionuclide in the body. Propagation of uncertainties is performed using Monte Carlo techniques.

Out of the five radionuclides, ^{137}Cs has the lowest uncertainty in the ingestion dose factors (Table 11.2.3, Figure 11.2.2). Cesium-137 distributes uniformly in the body; thus, there is little difference among the doses to various organs. Knowledge of the amount of potassium in the body of an exposed individual can lead to more precise dose estimation.

The ingestion dose factors and the associated uncertainties estimated for ^{90}Sr are presented in Table 11.3.3 and in Figure 11.3.2. Strontium-90 accumulates mostly in bone tissues. For a given intake, the dose to the bone surfaces or to the red bone marrow are almost three orders of magnitude larger than the dose to other tissues. Also, ^{90}Sr contributes significantly to the doses to the large intestine and rectum. Uncertainties are a factor of 2.2 for bone surface, 2.9 for red bone marrow, 3.5 for stomach and the small intestine, and 4.6 for all other organs.

Organ-specific ingestion dose factors for ^{60}Co are presented in Table 11.4.3 and Figure 11.4.3. Uncertainties in the organ doses are relatively large (a factor of 9.7) because of the large interindividual variation in the biokinetics of cobalt and in the absorption fraction (f_1) from the gastrointestinal tract to blood.

The biokinetics of ^{106}Ru are based solely on animal data. Ruthenium is poorly absorbed from the gastrointestinal tract. Immediately after ingestion, ruthenium tends to accumulate in the liver and kidneys, but over the long term it distributes uniformly in the body. The upper large intestine, colon, and rectum receive the largest doses from ingestion of ^{106}Ru. The uncertainty is small for the dose to the intestines (a factor of 2.8), and is larger for stomach (a factor of 6.6) and for all other organs (a factor of 8.8). The ingestion dose factors are presented in Table 11.5.5. and Figure 11.5.3.

Radioiodine accumulates in the thyroid gland, increasing the risk of thyroid disease. The dose factors from ingestion of ^{131}I are estimated for individuals of ages 0-15 (Table 11.6.3, Figure 11.6.6). A newborn ingesting ^{131}I receives a higher dose to the thyroid than any other children. The uncertainty in the thyroid dose for newborn is also the largest (a factor of 3.5). For other age groups, the uncertainty in the thyroid dose varies from a factor of 2.6 to a factor of 3.1. The largest contributors to the uncertainty in the dose to the thyroid are the thyroid mass and the fractional uptake of ^{131}I from blood to the thyroid gland.

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12.0 EXCESS LIFETIME RISK PER UNIT DOSE FOR CANCER INCIDENCE

12.1 Introduction

Section 12 presents the methodology used for obtaining the *excess lifetime risk* (*ELR*) of cancer incidence for a unit radiation dose received by people who may have been exposed to radionuclides released to White Oak Lake and the Clinch River from operations at the Oak Ridge National Laboratory. The *excess lifetime risk* is defined as the risk of acquiring radiation-induced cancer over a lifetime, in excess of the expected risk in an unexposed population[1] (background risk). The *excess lifetime risk per unit dose* is also called a *risk factor*. Additional definitions of terms used in this section are presented in Table 12.1.

Table 12.1 Terminology for risk estimates used in this report.

Notation	Name	Definition
	Risk	Probability (likelihood) of cancer incidence
	Risk estimate (projected risk)	Risk calculated for an exposed population by applying epidemiological data from atomic bomb survivors
B	Background cancer incidence rate	Cancer incidence rates in the state of Tennessee
ELR	Excess lifetime risk (or excess absolute risk)	The risk of acquiring radiation-induced cancer over a lifetime, above the background risk of cancer incidence
$ERR_{1\ Sv}$	Excess relative risk per unit dose	The risk of acquiring radiation-induced cancer over a lifetime, from a unit dose of radiation, normalized to the background risk of cancer incidence
$RF_{1\ Sv}$	Risk factor or Risk coefficient	Excess lifetime risk that is attributable to a unit dose of radiation

[1] Unexposed population = population not exposed to radionuclides released to the Clinch River (see Section 12.2.2.1).

The excess lifetime risk is estimated for adults exposed externally or internally to ^{137}Cs, ^{60}Co, ^{106}Ru, ^{90}Sr, ^{95}Zr, ^{95}Nb, and ^{144}Ce. Estimation of risk factors for these individuals is presented in Section 12.2. For intake of ^{131}I, children are the most at-risk individuals. Therefore, estimation of risk factors for children is presented separately in Section 12.3.

12.2 Excess Lifetime Risk Per Unit Dose for Adult Target Individuals

The dose-response relationship and the background incidence of cancer used in the estimation of the risk factors is based on age-adjusted values (Thompson et al., 1994). This approach is realistic for an individual growing and aging in the Clinch River area. Differences introduced by a different age-grouping or by consideration of an age-dependent analysis are addressed in Section 12.2.6.

12.2.1 Methodology used for estimation of the risk factors

The excess lifetime risk per unit dose of cancer from irradiation of a given organ was calculated based on epidemiological investigations of the survivors of the atomic bombings of Hiroshima and Nagasaki. The dose-response relationship from this study, as well as from other epidemiological studies, can be expressed on either a relative or an absolute basis (ICRP, 1991; NRC, 1990).

The **relative model** is based on the observation that, after a minimum latency period from the moment of first exposure, the risk of radiation-induced cancer will exhibit the same pattern as the background rate of cancer incidence. A relative model was inferred from many of the epidemiological data sets for most organs of interest. This model is also preferred by the International Commission on Radiological Protection (ICRP, 1991) and the National Research Council Committee on Biological Effects of Ionizing Radiation (BEIR V; NRC, 1990).

An alternative model is the **absolute** or **additive risk projection model**, which postulates that, after a given latency period, the excess risk of cancer will be independent of the background incidence rate. The additive model and the relative model should give similar results when they are applied to the population from which they were derived (NRC, 1990; Thompson et al., 1994). The main differences occur when they are used to estimate risks in populations that have different background incidence rates.

To apply the dose-response relationship from the Japanese A-bomb survivors to the U.S. population, a number of adjustment factors must be considered:

(a) bias correction due to uncertainty in the transfer of the dose-response relationship from the Japanese population to the U.S. population (b_{MOD});

(b) bias correction due to random errors in dosimetry (b_{RED});

(c) bias correction due to systematic errors in dosimetry (b_{SED});

(d) bias correction due to projection to lifetime (b_{PL}); and

(e) bias correction due to the difference between the response to a high dose rate (for A-bomb survivors) and exposure situations involving doses delivered over a longer period of time at a much lower rate (*DDREF* [2] correction) (b_{DDREF}).

Each of these bias correction factors is affected by uncertainty (Section 12.2.2), which must be taken into account in the calculation of risk. For the population exposed to radionuclides released to the Clinch River, dose-response equations based on the relative risk model were used. A corrected excess relative risk per unit dose ($\beta_{1\ Sv}$) was first determined by

$$\beta_{1Sv} = \beta_0 \cdot b_{MOD} \cdot b_{RED} \cdot b_{SED} \cdot b_{PL} / b_{DDREF} \qquad (12.1)$$

where β_0 = Age-adjusted excess relative risk per unit dose ($ERR_{1\ Sv}$).

Age-adjusted values of β_0 for cancer incidence were obtained from the extended Life Span Study (LSS-E85) of the A-bomb survivors cohort, as reported by Thompson et al. (1994) for all solid tumors and by Preston et al. (1994, cited by UNSCEAR, 1994) for leukemia (Table 12.2). The statistical or epidemiological error about β_0 was explicitly used in estimation of the uncertainty about the risk estimates.

The excess lifetime risk per unit dose for the U.S. population (risk factor; RF_{1Sv}; risk Sv^{-1}) is calculated as the product of the age-adjusted excess relative risk per Sv ($\beta_{1\ Sv}$) for cancer incidence and the age-adjusted annual background incidence rate of cancer (B) for a given cancer type, assuming an average 70-year lifetime:

$$RF_{1Sv} = \beta_{1Sv} \cdot B \cdot 70 \qquad (12.2)$$

[2] DDREF = Dose and Dose-Rate Effectiveness Factor.

Table 12.2 Excess relative risk per unit dose for cancer incidence ($ERR_{1\ Sv}$; risk Sv^{-1}) by cancer type for atomic bomb survivors (Thompson et al., 1994).

Cancer type	Mean	95% Confidence Interval	
		lower bound	upper bound
Total Solid Tumors	0.63	0.52	0.74
Oral Cavity	0.29	-0.09	0.93
Digestive System	0.38	0.25	0.52
Esophagus	0.28	-0.21	1
Stomach	0.32	0.16	0.5
Colon	0.72	0.29	1.28
Rectum	0.21	-0.17	0.75
Liver	0.49	0.16	0.92
Gallbladder	0.12	-0.27	0.72
Pancreas	0.18	-0.25	0.82
Respiratory System	0.8	0.5	1.2
Lungs	0.95	0.6	1.4
Skin (other than melanoma)	1	0.41	1.9
Female Breast	1.6	1.1	2.2
Uterus	-0.15	-0.29	0.1
Ovary	0.99	0.12	2.3
Prostate	0.29	-0.21	1.2
Urinary Organs and Kidneys	1.2	0.62	2.1
Urinary Bladder	1	0.27	2.1
Kidney	0.71	-0.11	2.2
Nervous System	0.26	-0.23	1.3
Thyroid	1.2	0.48	2.1
Leukemia[a]	4.4	3.2	5.5

[a] UNSCEAR, 1994

Risk factors are used to estimate the excess lifetime risk of cancer incidence for each organ by direct multiplication with the dose delivered to that particular organ (see Section 4).

12.2.2 Sources of Bias and Uncertainty

12.2.2.1 Background Incidence Rate of Cancer for the State of Tennessee

In this study, a population not exposed to radionuclides from the Oak Ridge reservation is called an "unexposed" population. This population may have a higher or a lower background incidence of cancer, depending at least in part on its exposure to other carcinogens. The *excess* risk is the risk from exposure

to radioisotopes from the reservation *in addition* to the background incidence of cancer in the unexposed population. According to the relative risk model (Eq. 12.2), the additional (or excess) risk is proportional to the background. Thus, the choice of the background is important in risk estimation.

The background incidence rate of cancer is similar across the State of Tennessee because of the similar lifestyles of the people. Only a very small part of the Tennessee population was exposed to radionuclides from the reservation by either living in the proximity of the reservation or by working in the nuclear facilities on the reservation. There are about 481,000 people living in Anderson, Roane, Loudon, and Knox counties, as opposed to about 4,851,000 in the entire state of Tennessee (1986-1993 data; Bashor, 1996). Given that not all the people in Anderson, Roane, Loudon, and Knox Counties have been exposed to the radioactivity released from the Oak Ridge Reservation to White Oak Creek and the Clinch River, the fraction of the Tennessee population exposed is less than 10%. Thus, the background incidence rates of cancer for the entire State of Tennessee are a good choice for this study. The risk estimates are, however, subject to a number of biases related to this choice of background (age at exposure, gender), and to a number of biases related to the choice of a relative risk model. These biases are discussed in the following sections.

In the State of Tennessee, 1989 was the first year when cancer incidence for all organs was reported by all appropriate hospitals. The background rates of cancer incidence were obtained from the Tennessee Department of Health (TDH) in Nashville in two data sets. The first set (TCRS, 1992), obtained in June 1996, contains 1989 to 1992 data organized as follows:

(a) the number of reported cases of cancer for Tennessee residents, by cancer type and age group, for both genders;

(b) the age-adjusted cancer incidence rates and the number of cases for each county in the State of Tennessee, for all cancer types, all races and both genders; and

(c) the age-adjusted cancer incidence rates for the entire State of Tennessee, for 23 cancer types, for each gender separately and a total for both genders.

The second data set (TCRS, 1996), obtained in August 1997, contains 1989 to 1994 data organized as follows:

(a) the age-dependent and age-adjusted cancer incidence rates, as well as the reported number of cases of cancer for Anderson, Blount and Roane counties in the State of Tennessee, for ten cancer types, for each gender separately and a total for both genders; and

(b) the age-dependent and age-adjusted cancer incidence rates, as well as the reported number of cases of cancer for all counties in Tennessee other than Anderson, Blount and Roane counties, for ten cancer types, for each gender separately and a total for both genders.

A number of important differences exist between the two data sets. The first difference is the coverage of the reporting area. Second, the number of organs for which cancer incidence is reported differs between the two data sets: the first data set is more complete, including the incidence rates for important cancer types such as leukemia and prostate (Figure 12.1). Third, the second set contains both the number of reported cancers and the associated population numbers, while the first data set contains only the number of cancers. The first data set is used in this report because it is more useful for estimating the risk of radiation-induced cancer incidence. However, the second set is used to analyze gender and age influence on the baseline cancer rates (Section 12.2.5).

The background cancer incidence rates used in calculations for the present report are age-adjusted, Tennessee-wide averages determined for both sexes in 1992 (Table 12.3). These data are considered to be representative for individuals living in the counties around the Oak Ridge Reservation who could have had significant exposures to radionuclides in water or in fish from the Clinch River.

None of the age-adjusted background incidence rates reported by the Tennessee Cancer Reporting System (TCRS, 1992) include confidence intervals obtained from statistical analysis of data other than the value for the total incidence of solid tumors (95% confidence interval). For the single values reported, a factor of 2 was used to calculate upper and lower bounds, and log-triangular distributions were assigned. An uncertainty factor of 2 produces a larger range of values than the observed confidence intervals for similar age-adjusted incidence rates (Washington State Cancer Registry; Miyahara, 1993). It is highly probable that this range encompasses the true but unknown value for the given background incidence rate. For the incidence rate for total solid tumors, a normal distribution was assigned based on the reported 95% confidence interval.

Table 12.3 Age-adjusted incidence rates[a] per 100,000 persons per year by cancer type for the State of Tennessee (TCRS, 1992).

Cancer Type	**Uncertainty Range**			**Type of Uncertainty**	**Distribution Type**
	Mean	**lower bound**	**upper bound**	**range**	
Total Solid Tumors	351.2	346.3	356.1	95% C.I.	normal
Oral Cavity And Pharynx	9.1	4.5	18.212	a factor of 2	log-triangular
Digestive System	N/A[b]	N/A	N/A		
Esophagus	3.4	1.7	6.8	a factor of 2	log-triangular
Stomach	6.2	3.1	12.4	a factor of 2	log-triangular
Colon	29.3	14.7	58.6	a factor of 2	log-triangular
Rectum	11.9	5.95	23.8	a factor of 2	log-triangular
Liver	1.9	0.95	3.8	a factor of 2	log-triangular
Gallbladder	N/A	N/A	N/A		
Pancreas	7.8	3.9	15.6	a factor of 2	log-triangular
Respiratory System	N/A	N/A	N/A		
Lungs	63.2	31.6	126.4	a factor of 2	log-triangular
Skin (other than melanoma)	N/A	N/A	N/A		
Female Breast	94.9	47.5	189.8	a factor of 2	log-triangular
Uterus	16.8	8.4	33.6	a factor of 2	log-triangular
Ovary	12.7	6.35	25.4	a factor of 2	log-triangular
Prostate	125.8	62.9	251.6	a factor of 2	log-triangular
Urinary Organs and Kidneys	N/A	N/A	N/A		
Urinary Bladder	13.4	6.7	26.8	a factor of 2	log-triangular
Kidney	7.9	3.95	15.8	a factor of 2	log-triangular
Nervous System	5.8	2.9	11.6	a factor of 2	log-triangular
Thyroid[c]	3.7	1.85	7.4	a factor of 2	log-triangular
Leukemia - Red Bone Marrow	7.5	3.8	15.0	a factor of 2	log-triangular

[a]The observed incidence rates were adjusted for the fraction of the population in each age group.
[b]N/A - not available
[c]1986 - 1993 data from Tennessee Department of Health (TCRS, 1996)

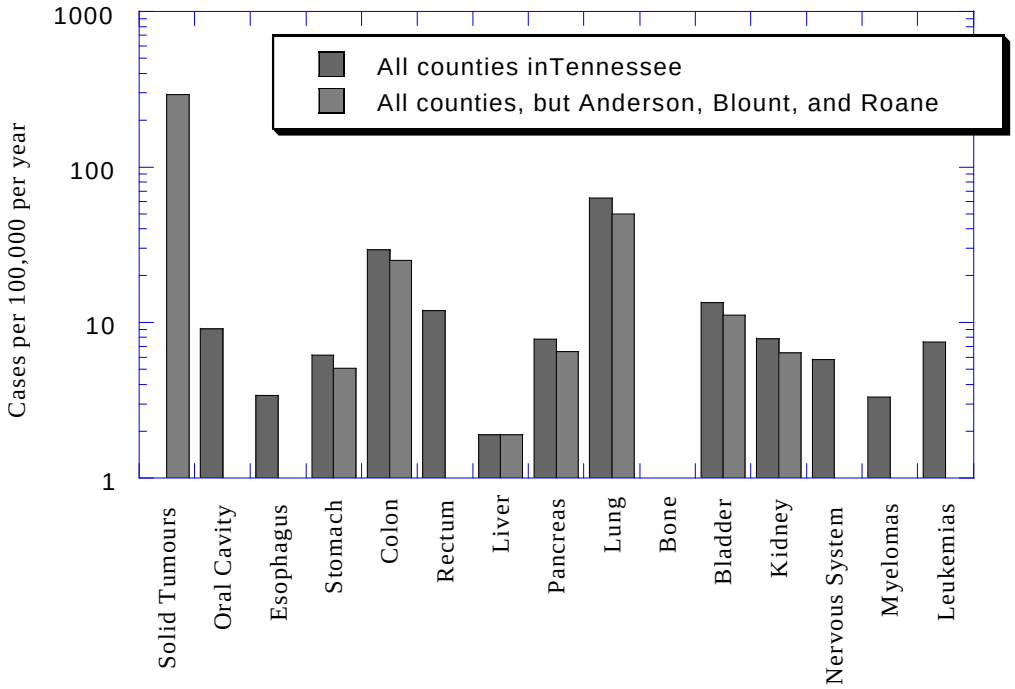

Figure 12.1 Age-adjusted cancer incidence rates for the State of Tennessee for both genders in 1992. The rates are adjusted to the 1970 U.S. standard population.

12.2.2.2 Uncertainty in the Transfer of the Dose-Response Relationship from the Japanese Population to the U.S. Population

Several approaches can be used to apply the dose-response relationship obtained from the A-bomb survivors to a different population with a different exposure situation. The first approach is a **multiplicative** or **relative transfer** model (ICRP, 1991). This model is based on the assumption that the ratio between the excess risk and the background incidence is, at any age, invariant over populations with different background incidence rates. That is, the excess relative risk per unit dose ($ERR_{1\ Sv}$) derived for the Japanese population and applied to the U.S. background incidence of cancer is assumed to give a reasonable approximation of the excess risk due to exposure to radiation for the U.S. population.

An alternative model for estimating the risk of radiation-induced cancer in the U.S. population based on the risk coefficients determined for the Japanese cohort is the **additive transfer model** (ICRP, 1991). This model assumes that the absolute risk of cancer can be estimated directly for the U.S. population through simple addition to the natural background incidence. There is insufficient evidence at present to conclude that one model is clearly better than the other for transferring the estimate of risk from data obtained from the Atomic Bomb Survivors Study. Even though ICRP emphasizes the relative transfer model (ICRP, 1991), the Commission derives its risk coefficients from an arithmetic average of the two models. The Environmental Protection Agency (Puskin and Nelson, 1995) uses risk coefficients based on the geometric mean of the two models (except for liver, for which a relative transfer model was preferred, and bone cancer, for which an absolute transfer model was chosen).

In this study, three alternative models have been considered to apply the A-bomb survivors dose-response relationship to the U.S. population: (a) a relative risk model, (b) an absolute risk model, and (c) a geometric mean model (based on the geometric mean of the excess lifetime risks estimated using the relative and absolute risk models). The structure of our calculations (Eq. 12.1 - 12.2) was first designed using a relative risk model. Then the uncertainty introduced by the choice of transfer model was explicitly represented by a multiplicative bias correction (b_{MOD}) defined by the following equation:

$$b_{MOD} = 1 \qquad \textit{for the relative risk transfer model} \tag{12.3a}$$

or

$$b_{MOD} = \frac{ELR_{1\ Sv}^{Absolute\ Risk}}{ELR_{1\ Sv}^{Relative\ Risk}} \qquad \textit{for the absolute risk transfer model} \tag{12.3b}$$

or

$$b_{MOD} = \sqrt{\frac{ELR_{1\ Sv}^{Absolute\ Risk}}{ELR_{1\ Sv}^{Relative\ Risk}}} \quad for\ the\ geometric\ mean\ risk\ transfer\ model \qquad (12.3c)$$

where $ELR_{1\ Sv}^{Relative\ Risk}$ and $ELR_{1\ Sv}^{Absolute\ Risk}$ are the deterministic estimates of the excess lifetime risk per unit of exposure based on a relative and an absolute risk projection model, respectively. They are calculated, assuming an average 70-year lifetime, as follows:

$$ELR_{1\ Sv}^{Relative\ Risk} = \beta_{1\ Sv} \cdot B \cdot 70\ years \qquad (12.4)$$

and

$$ELR_{1Sv}^{AbsoluteRisk} = EAR_{1PYSv} \cdot 70 years \qquad (12.5)$$

where:

$\beta_{1\ Sv}$ = the excess relative risk per unit dose [ERR Sv^{-1}] (Thompson et al., 1994) (Table 12.2);

B = the background incidence rate of cancer [yr^{-1}] for the region of interest (TCRS, 1996) (Table 12.3);

$EAR_{1\ PY\ Sv}$ = the excess absolute risk per person-year per sievert [EAR $(PY\ Sv)^{-1}$] (Thompson et al., 1994) (Table 12.4).

The correction factor (b_{MOD}) is described as a discrete random variable, for which the value calculated for a given transfer model is sampled by a Monte Carlo procedure with a frequency equal to a subjective weight (Table 12.5) assigned to that particular model. In this fashion, the resulting distributions for the risk estimates contain the uncertainty due to the selection of the most appropriate model for transferring the dose response from the A-bomb survivors to the U.S. population.

Table 12.4 Excess absolute risk per 10,000 persons per unit dose for cancer incidence [EAR $(PY\ Sv)^{-1}$; EAR per person-year per sievert] by cancer type for atomic bomb survivors (Thompson et al., 1994).

Cancer type	**Mean**	**Uncertainty Range 95% Confidence Interval**	
		lower bound	**upper bound**
Total Solid Tumors	29.7	24.7	34.8
Oral Cavity	0.23	-0.08	0.65
Digestive System	10.4	7.0	14.0
Esophagus	0.3	-0.23	1.0
Stomach	4.8	2.5	7.4
Colon	1.8	0.74	3.0
Rectum	0.43	-0.35	1.5
Liver	1.6	0.54	2.9
Pancreas	0.24	-0.36	1.1
Respiratory System	4.4	2.9	6.1
Lungs	4.4	2.9	6.0
Skin (other than melanoma)	0.84	0.4	1.4
Female Breast	6.7	4.9	8.7
Uterus	-1.1	-2.1	0.68
Ovary	1.1	0.15	2.3
Prostate	0.61	0.15	2.3
Urinary Organs and Kidneys	2.1	1.1	3.2
Urinary Bladder	1.2	0.34	2.1
Kidney	0.29	-0.5	0.79
Nervous System	0.19	-0.17	0.81
Thyroid	1.6	0.78	2.5
Leukemia[a]	N/A	N/A	N/A

[a] N/A = not available

Table 12.5 Bias correction factors (b_{MOD}) and their relative weights for the different models used to apply the Atomic Bomb survivors dose-response relationship to the U.S. population. The correction factors are based on deterministic estimates (Eq. 12.3; 12.4; 12.5) of the excess lifetime risk from exposure to 1 Sv, obtained using a relative or an absolute risk model.

Cancer Type	Excess Lifetime Risk from 1 Sv		Relative Risk Transfer Model		Geometric Mean Transfer Model[a]		Absolute Risk Transfer Model	
	Relative Risk Model	Absolute Risk Model	b_{MOD}	Weight	b_{MOD}	Weight	b_{MOD}	Weight
Total Solid Tumors[b]	1.8E-01	2.1E-01						
Oral Cavity	1.2E-03	1.6E-03	1.0	40%	1.2	40%	1.3	20%
Esophagus	1.3E-03	2.1E-03	1.0	40%	1.3	40%	1.6	20%
Stomach	1.2E-03	3.4E-02	1.0	70%	5.4	25%	29.0	5%
Colon	1.6E-02	1.3E-02	1.0	40%	0.90	40%	0.81	20%
Liver	9.3E-04	1.1E-02	1.0	40%	3.5	40%	12.0	20%
Pancreas	1.3E-03	1.7E-03	1.0	40%	1.1	40%	1.3	20%
Lung	4.0E-02	3.1E-02	1.0	40%	0.87	40%	0.76	20%
Skin[b]								
Female Breast	1.3E-01	4.7E-02	1.0	10%	0.59	30%	0.35	60%
Uterus[c]								
Prostate	4.3E-02	4.3E-03	1.0	40%	0.32	40%	0.10	20%
Ovary	1.0E-02	7.7E-03	1.0	40%	0.87	40%	0.76	20%
Kidneys	3.0E-03	2.0E-03	1.0	40%	0.83	40%	0.68	20%
Bladder	1.1E-02	8.4E-03	1.0	40%	0.86	40%	0.74	20%
Thyroid[d]	5.7E-03	1.1E-02	1.0	100%				
Leukemia[d]	3.7E-02		1.0	100%				

[a]Based on the geometric mean of the absolute and relative risk.
[b]Risk estimates were not obtained for these cancer types (see Table 12-12).
[c]The uterus has a negative dose-response relationship (Thompson et al., 1994); risk was not estimated for the uterus.
[d]The relative risk model was the only transfer model considered appropriate for estimation of cancer risk for these cancer types.

For leukemia, ICRP (1991) has shown that the choice of a model has little influence on the projection of the lifetime risk from the A-bomb survivors to other populations. Therefore, for leukemia, the relative risk model is given preference in our investigation and was assigned a weight of 100%. A weight of 100% was also assigned to the relative risk model for estimating thyroid cancer based on the abundant data summarized by Ron et al. (1995).

UNSCEAR (1994) states that, in the case of stomach cancer, "the relative risks may be more similar in populations with disparate background stomach cancer rates." The relative risk is defined as the ratio of total risk (sum of background and excess risk) to the background risk. Thus, for stomach cancer, a 70% weight was given to the relative risk model; however, to account for uncertainty, a 5% weight was assigned to the absolute risk model, and a 25% weight to the geometric mean model. For breast cancer, the absolute risk model was assigned a weight of 60%, the relative risk model was assigned a 10% weight, and the geometric mean model was assigned a 30% weight. The assignment of these weights was made after obtaining additional insight from discussions with Dr. Warren Sinclair on the transfer of information on the risk of breast cancer across populations (Sinclair, 1997), and a review of a recent publication by Dr. Charles E. Land on cancer and radiation dose among the Japanese survivors of the bombing of Hiroshima and Nagasaki (Land, 1995). For all other cancers, the relative risk model and the geometric mean model were each assigned a 40% weight, while the absolute risk model received the remaining 20% (Table 12.5).

12.2.2.3 Bias Due to Random Error in Doses Assigned to Individual A-bomb Survivors.

Values of the excess relative risk per unit dose ($ERR_{1\ Sv}$) are based on the average cohort dose estimates for the A-bomb survivors produced by the Radiation Effects Research Foundation (RERF) using the 1986 Dosimetry System (DS86; Roesch, 1987). However, the individual radiation dose estimates are affected by substantial uncertainties. These uncertainties, known as "random" errors in dosimetry (Pierce et al., 1990), are attributable to, for instance, the difficulty of precisely determining either the position of the individual at the time of the bombing or the shape and composition of any shielding material. When random errors associated with the individual doses are considered, the averages of the estimated cohort doses are found to be lower than the original averages in which no errors were considered. Moreover, the larger the dose, the larger the error, and the larger the bias toward lower averages (Pierce et al., 1990). Consequently, the adjusted slope of the linear dose-response function ($ERR_{1\ Sv}$) that accounts for the presence of random errors in the doses estimated for each individual in the cohort is greater than the original slope (i.e., the risk for a given dose is higher). Explicit consideration of the random errors in these dose estimates leads to a positive bias in the estimated relative risk.

The magnitude of the difference between the adjusted and original values of $ERR_{1\ Sv}$ depends on the type of distribution used to describe the error in an individual dose. Pierce et al. (1996) studied the effects of lognormal distributions and on the magnitude of the error. Lognormal or normal distributions or a combination of the two distributions have been used. The dosimetric random errors were estimated at 30-40% (Jablon, 1971, cited by Pierce et al., 1990), and therefore, coefficients of variation of 30-40% were assigned to the distribution. Pierce et al. (1990) used four types of distributions (also called error models): lognormal with 30% error, lognormal with 40% error, normal with 40% error, and a combination of two

lognormal distributions with 40% error. For the three lognormal models, a positive bias in the $ERR_{1\,Sv}$ of 6.8-11.4% was obtained for all cancers except leukemia; a range of 4.3-7.2% was obtained for leukemia.

In an analysis of the proportion of cells with chromosome aberrations in cultured blood lymphocytes from A-bomb survivors, Sposto et al. (1991) revealed that the dose-response relationship for individuals who reported severe epilation (hair loss) is steeper than for individuals who did not report severe epilation. This effect could be explained by the presence of random dosimetry errors in the range of 45 to 50% (NCRP, 1997; Grogan et al., 1997). This range for random errors extends the range proposed by Jablon (1971), so that the bias in the estimated relative risk could be even larger than the bias estimated by Pierce et al. (1990). The uncertainty in the bias due to random errors (b_{RED}) adopted in this study is similar to that proposed by Grogan et al. (1997) and is described as a uniform distribution (Table 12.6) with a range of 1.04 to 1.09 for leukemia and 1.06 to 1.14 for all other cancer types or organ systems.

Table 12.6 Adjustment factors for bias due to random errors in dosimetry (b_{RED}).

Cancer type	Min	Max	Distribution Type	References
Leukemia	1.04	1.09	uniform	Pierce et al. (1990); Grogan et al. (1997)
All other cancers	1.06	1.14	uniform	Pierce et al. (1990); Grogan et al. (1997)

12.2.2.4 Bias Due to Systematic Errors in Doses Assigned to Individual A-bomb Survivors

External exposures of the A-bomb survivors are given by two major components: gamma rays and neutrons of various energies. The total dose to a given organ is calculated by adding the gamma-ray dose to the neutron dose weighted by the relative biological effectiveness (*RBE* [3]) of neutrons. The uncertainties in the gamma-ray dose, neutron dose or *RBE* that apply to a large proportion of the exposed cohort are known as "systematic" errors in dosimetry.

The first source of uncertainty is a systematic underestimation of the gamma ray free field[4] (in unshielded areas) for Hiroshima survivors (NCRP, 1997). This source of bias was negligible near the hypocenter[5],

[3]RBE = Relative Biological Effectiveness - a factor describing the sensitivity of the biological material to different types of radiation and depending on the physical nature of the radiation field, the type of biological material and the particular biological response considered. The larger the RBE the more effective the radiation is. Gamma rays are considered to have RBE = 1.

[4]Gamma ray free field irradiation is the irradiation received in an unshielded area from early or prompt gamma rays emitted during the atomic explosion and from delayed gamma rays emitted by the fission products.

[5]The hypocenter is the point on the surface of the earth directly above or below the point at which an atomic or hydrogen bomb explodes.

but it increased significantly with distance from the hypocenter (about 20% at 1400 meters). The effect of this dose underestimation can be translated into a negative bias in the estimates of excess relative risk (a reduction factor of about 1 to 1.4 with a most probable value of 1.1; NCRP, 1997). This bias applies only for Hiroshima; the dose estimates for Nagasaki are thought to be accurate (NCRP, 1997). Since 67.6% of the study cohort was exposed at Hiroshima (Thompson et al., 1994), the $ERR_{1\ Sv}$ should be adjusted by about 5%. A multiplicative parameter ($b_{SED\tilde{a}}$) with a triangular distribution (min = 0.9, likeliest = 0.95, max = 1.0) was assigned to account for the errors in the component of the total dose contributed by gamma rays (Table 12.7).

Table 12.7Adjustment factors for bias due to systematic errors in dosimetry (b_{SED}).

Source of uncertainty	Cancer type	Central Value	Min	Max	Distribution Type	References
$b_{SED\tilde{a}}$	All cancer sites	0.95	0.90	1.00	triangular	this report
$b_{SED\ RBE}$	Leukemia	1.0	0.85	1.15	triangular	this report
	All other sites	1.0	0.90	1.10	triangular	this report
$b_{SED\ neutron}$	All cancer sites	1.15	1.0	1.30	triangular	NCRP (1997)

The second source of uncertainty is the selected value for the *RBE* of neutrons. Values proposed for the *RBE* of neutrons vary from 1 to 20. The use of an *RBE* of 10 will decrease the calculated risk by 10% for leukemia and 12% for lung cancer (Grogan et al., 1997), as compared to the risk obtained from doses having an RBE of 1. NCRP (1997) gives an expected decrease of 10% for leukemia and 6% for all other cancers. According to Preston et al. (1992-1993) as cited by Grogan et al. (1997), a decrease of 22% is obtained for all cancers except leukemia, assuming an RBE of 20, and a decrease of 13% for an RBE of 10.

The values of $ERR_{1\ Sv}$ from Thompson et al. (1994) are based on the latest Hiroshima and Nagasaki dosimetry (DS86), which uses an RBE of 10. Therefore, no bias due to *RBE* should be expected. However, since the real *RBE* for neutrons could have been between 1 and 20, according to the neutron energy, an uncertainty factor should still be considered.

The following multiplicative adjustment factor ($b_{SED\ RBE}$) for systematic errors introduced by the *RBE* was selected (Table 12.7):

(a) for all cancers - a triangular distribution *(min = 0.9, likeliest = 1.0, max = 1.1)*. This includes a 10% error introduced by selecting different values of RBE around 10 (between 1 and 20).

(b) for leukemia - a triangular distribution *(min = 0.85, likeliest = 1.0, max = 1.15)*. This includes a 15% error introduced by the RBE variation around 10 (between 1 and 20).

The third source of systematic error is introduced by the uncertainty in the magnitude of the neutron component of the total dose. NCRP (1997) analyzes the reasons for this uncertainty. The neutron flux and the subsequent dose were evaluated theoretically and verified against measurements of ^{60}Co activation in steel and measurements with ^{36}Cl. The calculated neutron flux in Nagasaki was in agreement with the measurements. For Hiroshima, however, the measurements are in agreement with the calculations only at positions close to the hypocenter; at a distance of about 1600 meters, the measurements indicate an activation larger than that calculated by a factor of 10. NCRP (1997) summarized the effects of this bias and suggested that it should reduce the risk by a factor ($b_{SED\ neutron}$) of about 1.15. The distribution suggested by NCRP (1997) for $b_{SED\ neutron}$ (triangular with min = 1.0, likeliest = 1.15, max = 1.3) was adopted here (Table 12.7).

Finally, the total bias due to systematic errors in dosimetry was calculated as

$$b_{SED} = \frac{b_{SED\,\gamma} \cdot b_{SEDRBE}}{b_{SED\ neutron}} \qquad (12.6)$$

12.2.2.5 Bias Due to the Projected Lifetime of the Japanese Survivors Cohort

The values of $ERR_{1\ Sv}$ for cancer incidence are determined based on the Atomic Bomb Survivors cohort. Many of the survivors are still alive, and thus not all of the radiation-induced cancers have yet been manifested. As the number of years after exposure increases, both the total number of cancers in the exposed population and the natural background incidence of cancer in the unexposed population will increase. As the study cohort ages, the excess relative risk will decrease because of the large increase in cancers produced by natural causes. To date, this effect of attained age has been shown for the study cohort for cancer incidence (Thompson et al., 1994) and for cancer mortality (Pierce et al., 1996) for different ages at exposure. The effect is strong for people who were children at the time of bombing and very low for adults exposed at ages over 40. By extrapolating the curves in Figure 4 of Thompson et al. (1994), one can conclude that the values for excess relative risk per Sv, which are presently derived from 30 years (1958-1987) of follow-up (Thompson et al., 1994), will decrease as the cohort is followed further. This decrease is called the "effect of projection to lifetime." However, the magnitude of the decrease is uncertain.

The values of $ERR_{1\ Sv}$ used in this report must be reduced by a fraction that accounts for the effects of projection to lifetime. To estimate the reduction fraction, Grogan et al. (1997) analyzed three models proposed by UNSCEAR (1994) for projection of risk to lifetime. All three models assume a 10-year latency period for cancer development after a single irradiation. The first model assumes that the risk of cancer remains constant over time; that is, there is no effect due to projection to lifetime. The second and third models assume that the risk is constant for the first 45 years after exposure and decreases linearly

thereafter. In the second model, the risk decreases to the average risk for an individual who was 50 years old at exposure, and in the third model, the risk decreases linearly to zero at an attained age of 90. The differences in predictions made by these models for different ages at time of exposure give an indication of the range of values to be considered for the reduction factor due to the projection to lifetime.

The largest departure from the predictions made by a constant $ERR_{1\,Sv}$ was obtained with the third model (UNSCEAR, 1994). For an exposure to a newborn, a reduction of lifetime risk by 51% was predicted by the third model, while the second model predicted a reduction of 35%. For an exposure after 50 years of age, no differences were found among the different models. The second model predicted a reduction of 9% in the average for all ages at exposure, while the third model predicted a reduction of 20%.

For this analysis, the reduction factor for the projection of risk to lifetime was obtained from the third model. This choice is in agreement with the reduction factor proposed by Grogan et al. (1997). A triangular distribution (min = 0.5, mode = 0.8, max = 1.0) was assigned to this parameter for all cancers other than leukemia (Table 12.8).

Table 12.8 Adjustment factors for bias due to projection to lifetime (b_{PL}).

Cancer Type	Central Value	Min	Max	Distribution Type	References
Leukemia	1			no uncertainty	Grogan et al. (1997)
All other sites	0.8	0.5	1.0	triangular	UNSCEAR (1994); Grogan et al. (1997)

The temporal behavior of leukemia is rather complicated, but essentially all expected excess leukemias have been observed (Grogan et al., 1997). Thus, the risk estimate appears to be complete for the A-bomb survivor cohort, and no modification is required for projection to lifetime (Table 12.8).

12.2.2.6 Dose and Dose-Rate Reduction Factor Adjustment

The A-bomb survivors were exposed to a single dose of radiation delivered at a very high rate. Based on abundant radiobiological information, it has been well established that a low-LET dose of radiation delivered at a low dose rate (or even in fractionated doses) should be less effective than the same total dose at high dose rates (UNSCEAR, 1994). Human data, however, are more limited, and the uncertainty introduced by this limitation must be taken into consideration when a risk of cancer is predicted.

To estimate the response at low dose rates, the $ERR_{1\ Sv}$ obtained from A-bomb survivors is reduced by a Dose and Dose Rate Effectiveness Factor (DDREF). ICRP (1991) recommends a DDREF of 2, a value which is also accepted by NCRP (1993). UNSCEAR (1993) summarizes experimental information on the dose-response relationship for different organs and recommends DDREF values of 2 for leukemia, 1.7 for all cancers combined, 2 for solid cancers, and 3 for thyroid cancer. A range of 1 to 5 and a most likely value of 2 for the DDREF have been suggested from a study of the uncertainties in risk estimates (for fatal cancers) used in radiation protection (NCRP, 1997). Distributions to describe the uncertainty in the DDREF for leukemia, lung, liver, bone, and total cancers have also been proposed by Dr. W. Sinclair in his work for dose reconstruction at Rocky Flats, Colorado (Grogan et al., 1997).

The distributions used in this report (Table 12.9) are based on those used by Grogan et al. (1997) and on an informal discussion with experts outside of this project team on the probability that the DDREF might be exactly 1.0. These experts were Dr. W. Sinclair (president emeritus, NCRP), Dr. P. Groer (Dept. of Nuclear Engineering, Univ. of Tennessee), Dr. J. Puskin (EPA), and Mr. C.B. Nelson (EPA). Based on these discussions, subjective weights were assigned to a DDREF of 1.0 (Table 12.9), with the remaining weight assigned to a continuous piecewise uniform distribution ranging from 1.0 to 5.0 for most cancers and from 1.0 to 4.0 for total solid tumors. Triangular distributions were assigned for leukemia (min = 1, likeliest = 2, max = 7) and for lung (min = 2, likeliest = 4, max = 10).

Table 12.9 Dose and dose-rate reduction factors (b_{DDREF}).

Cancer type	Central Value	Min	Max	Distribution Type	References
Lung and respiratory system	4	2	10	triangular	Grogan et al. (1997)
Leukemia	2	1	7	triangular	Grogan et al. (1997)
	Range or Value		Relative Probability		
Solid Tumors	1		15%	piecewise	this report
	1.0 - 1.5		25%	uniform	
	1.5 - 2.0		30%		
	2.0 - 3.0		20%		
	3.0 - 4.0		10%		
All other sites	1		15%	piecewise	this report
	1.0 - 1.5		20%	uniform	
	1.5 - 2.0		30%		
	2.0 - 3.0		20%		
	3.0 - 4.0		10%		
	4.0 - 5.0		5%		

12.2.3 Estimates of the Risk Factors

The Atomic Bomb Survivors data (Table 12.2) indicate leukemia as the highest risk from exposure to high doses and dose rates, followed by breast cancer, thyroid cancer, and cancer of the bladder and other urinary organs. The ranking of the adjusted $ERR_{1\,Sv}$ (Table 12.10) is different from the original $ERR_{1\,Sv}$ (leukemia, followed by liver cancer, thyroid cancer, and non-melanoma of skin). The reason for the change in ranking comes from the differences between the lifetime risk obtained by using a relative risk model and an absolute risk model (Table 12.5). The largest difference is found for liver cancer. The background incidence of liver cancer is very low in the U.S. as compared to Japan, and thus, the lifetime risk based on a relative risk model is much lower than the lifetime risk based on an absolute risk model. The reverse situation occurs for breast cancer, for which the U.S. background incidence is much higher than in Japan. Consequently, the lifetime risk of breast cancer derived using a relative model is much higher than the lifetime risk using an absolute risk model.

To obtain the risk factors (RF_{1Sv}), the adjusted $ERR_{1\,Sv}$ are multiplied by the background incidence rate of cancer for the Tennessee population (Table 12.11), because of differences in the background incidence rates of cancer. The highest result was obtained for the risk factor that relates the incidence of total solid tumors to a unit of whole body dose. Breast cancer and leukemia are still the most likely cancers from a unit dose of irradiation, but prostate, colon, and lung cancers also have high risk factors because of their high natural incidence.

Although the $ERR_{1\,Sv}$ were available for 23 organ systems or cancer sites, the risk factors were calculated only for the 18 for which the background incidence of cancer was available (Table 12.12). The uterus has a negative dose-response relationship (Thompson et al., 1994), and thus the risk factor was not estimated for the uterus.

12.2.4 Sensitivity Analysis

A sensitivity analysis was performed to identify the most important contributors to the uncertainty in the risk factors used in the estimation of risks for selected cancer types: leukemia, breast and colon cancer (Figure 12.2). The dose and dose-rate reduction factor (b_{DDREF}) contributed close to 50%, being the major source of uncertainty for all three types of cancer. The sensitivity of the colon to radiation, expressed as the excess relative risk per unit dose ($ERR_{1\,Sv}$), is the second major contributor to the uncertainty in the risk of colon cancer, while, for leukemia and breast cancer, the background cancer incidence rate is the second contributor to the total uncertainty.

Table 12.10 Values for the Excess Relative Risk per unit dose ($ERR_{1\ Sv} = \beta_{1\ Sv}$) from Table 12.2, adjusted for potential biases.

Cancer Type	**Excess Relative Risk[a]**		
	lower bound	**Best Estimate**	**upper bound**
Total Solid Tumors	0.12	0.31	0.59
Oral Cavity	0.01	0.07	0.52
Digestive System	0.05	0.15	0.34
Esophagus	0.01	0.10	1.11
Stomach	0.04	0.16	2.39
Colon	0.08	.025	0.77
Rectum	0.01	0.05	0.38
Liver	0.07	0.47	5.86
Gallbladder	0.00	0.02	0.33
Pancreas	0.00	0.04	0.51
Respiratory System	0.04	0.10	0.24
Lungs	0.05	0.11	0.27
Skin (other than melanoma)	0.10	0.36	1.10
Female Breast	0.10	0.35	0.91
Uterus	-0.12	-0.06	-0.02
Ovary	0.07	0.31	1.31
Prostate	0.00	0.03	0.32
Urinary Organs and Kidneys	0.10	0.35	1.05
Urinary Bladder	0.09	0.33	1.16
Kidney	0.02	0.15	1.06
Nervous System	0.00	0.04	0.60
Thyroid	0.13	0.44	1.27
Leukemia	0.61	1.22	2.93

[a]The best estimate is the 50th percentile of the uncertainty range. The lower bound and the upper bound are the 2.5th and 97.5th percentiles, respectively, of the uncertainty range.

Table 12.11 Risk factors for cancer incidence by cancer site or organ system[a] for the State of Tennessee ($RF_{1\ Sv}$; risk Sv^{-1}).

Cancer Type	**Risk Factors[b] (risk Sv^{-1})**		
	lower bound	**Best Estimate**	**upper bound**
Total Solid Tumors	2.9×10^{-2}	7.5×10^{-2}	1.5×10^{-1}
Oral Cavity	5.8×10^{-5}	4.4×10^{-4}	3.9×10^{-3}
Digestive System			
Esophagus	2.5×10^{-5}	2.4×10^{-4}	2.7×10^{-3}
Stomach	1.8×10^{-4}	7.1×10^{-4}	1.1×10^{-2}
Colon	1.3×10^{-3}	4.9×10^{-3}	1.8×10^{-2}
Rectum	3.6×10^{-5}	4.2×10^{-4}	3.7×10^{-3}
Liver	6.9×10^{-5}	6.3×10^{-4}	9.3×10^{-3}
Gallbladder			
Pancreas	1.4×10^{-5}	2.4×10^{-4}	2.5×10^{-3}
Respiratory System			
Lungs	1.7×10^{-3}	5.0×10^{-3}	1.3×10^{-2}
Skin (other than melanoma)			
Female Breast	6.4×10^{-3}	2.3×10^{-2}	6.8×10^{-2}
Uterus			
Ovary	5.1×10^{-4}	2.8×10^{-3}	1.4×10^{-2}
Prostate	1.7×10^{-4}	2.5×10^{-3}	3.5×10^{-2}
Urinary Organs and Kidneys			
Urinary Bladder	8.0×10^{-4}	3.0×10^{-3}	1.1×10^{-2}
Kidney	9.5×10^{-5}	8.0×10^{-4}	5.6×10^{-3}
Nervous System	1.1×10^{-5}	1.9×10^{-4}	2.7×10^{-3}
Thyroid	3.3×10^{-4}	1.1×10^{-3}	3.8×10^{-3}
Leukemia	2.5×10^{-3}	6.4×10^{-3}	1.8×10^{-2}

[a] Risk factors could not be calculated for all cancer types due to the absence of complete information (see text).

[b] The best estimate is the 50th percentile of the uncertainty range. The lower bound and the upper bound are the 2.5th and 97.5th percentiles, respectively, of the uncertainty range.

Table 12.12 Summary of the types of information available for calculations and of the calculations performed, by cancer type.

Cancer Type	Types of Information		
	$ERR_{1\ Sv}$ [a]	B[b]	Estimated Risk Factors
Total Solid Tumors	C	C	C
Oral Cavity	C	C	C
Digestive System	C		
Esophagus	C	C	C
Stomach	C	C	C
Rectum	C	C	C
Liver	C	C	C
Gallbladder	C	C	C
Pancreas	C	C	C
Respiratory System	C		
Lungs	C	C	C
Skin (other than melanoma)	C		
Female Breast	C	C	C
Uterus	C [c]	C	
Ovary	C	C	C
Prostate	C		C
Urinary Organs and Kidneys	C		
Urinary Bladder	C	C	C
Kidney	C	C	C
Nervous System	C	C	C
Thyroid	C	C	C
Leukemia-Red Bone Marrow	C	C	C
Totals	23	18	18

[a] Excess relative risk per unit dose from Atomic Bomb survivors study (Thompson et al., 1994).
[b] Background incidence of cancer for the State of Tennessee.
[c] The uterus has a negative dose-response relationship (Thompson et al., 1994); risk was not estimated for the uterus.

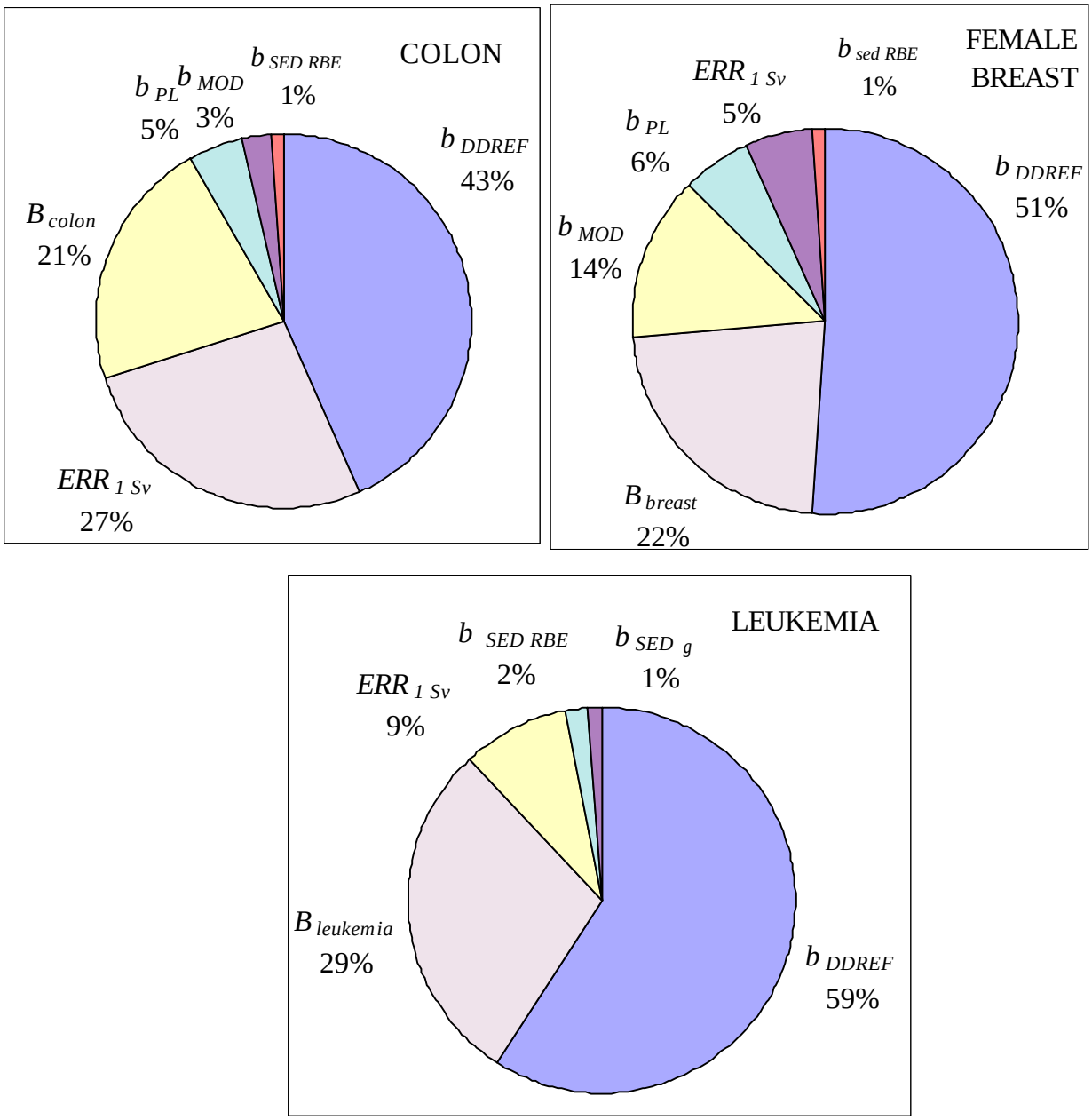

Figure 12.2 Most important contributors to the uncertainty in the risk factors for selected cancer types.

Breast cancer is rarer in Japan than in the United States. Also, estimates of risk for the U.S. population based on a relative risk transfer model are very different from those based on an absolute risk transfer model (Table 12.4). The uncertainty introduced by the choice of the transfer model (b_{MOD}) is of dominant importance for breast cancer.

At least for these types of cancer, a better understanding of the differences between the response of the human body to low versus high dose rates of radiation will improve the accuracy of the risk estimates. For leukemia and breast cancer, obtaining more accurate background cancer incidence rates will reduce the uncertainty in the risk estimates. Finally, for breast cancer, it is essential to learn the type of model that is most appropriate for applying the Japanese survivors data to the U.S. population.

12.2.5 Influence of Gender and Age-at-Exposure in Risk Estimates

Risk of cancer incidence is estimated in this study based on the excess relative risk per unit dose for both genders and all ages from the Atomic Bomb Survivors Study applied to the background incidence of cancer for both genders and all ages for the entire State of Tennessee. This section discusses how the current risk estimates would change if a gender and age-specific analysis were employed. Such an analysis is work-intensive, and it should be performed only if the risk estimates produced by this study are large enough to warrant further refinements.

For the particular case of exposure to ^{131}I released from Oak Ridge Reservation, the risk of radiation-induced thyroid cancer is estimated for each gender and each age group from newborn to age 15. Details on this gender- and age-specific analysis are presented in Section 12.3.

12.2.5.1 Influence of Gender

Gender is not an issue in risk estimation for gender-specific organs such as breast, uterus, and ovary for females, or prostate for males. For other organs, both the background cancer incidence rates and the excess relative risk coefficients from the Atomic Bomb Survivors from Hiroshima and Nagasaki are gender-dependent parameters.

In Tennessee, the incidence of leukemia is higher for males than for females (Figure 12.3), but the differences are not large. Using gender-specific cancer incidence rates, instead of an adjusted rate, the risk estimate for leukemia would have only minor variations (about 30%). The situation is similar for other organs, such as colon (10% to 20%), nervous system (about 10%), and pancreas (about 10%). Differences in incidence rates are important for lungs

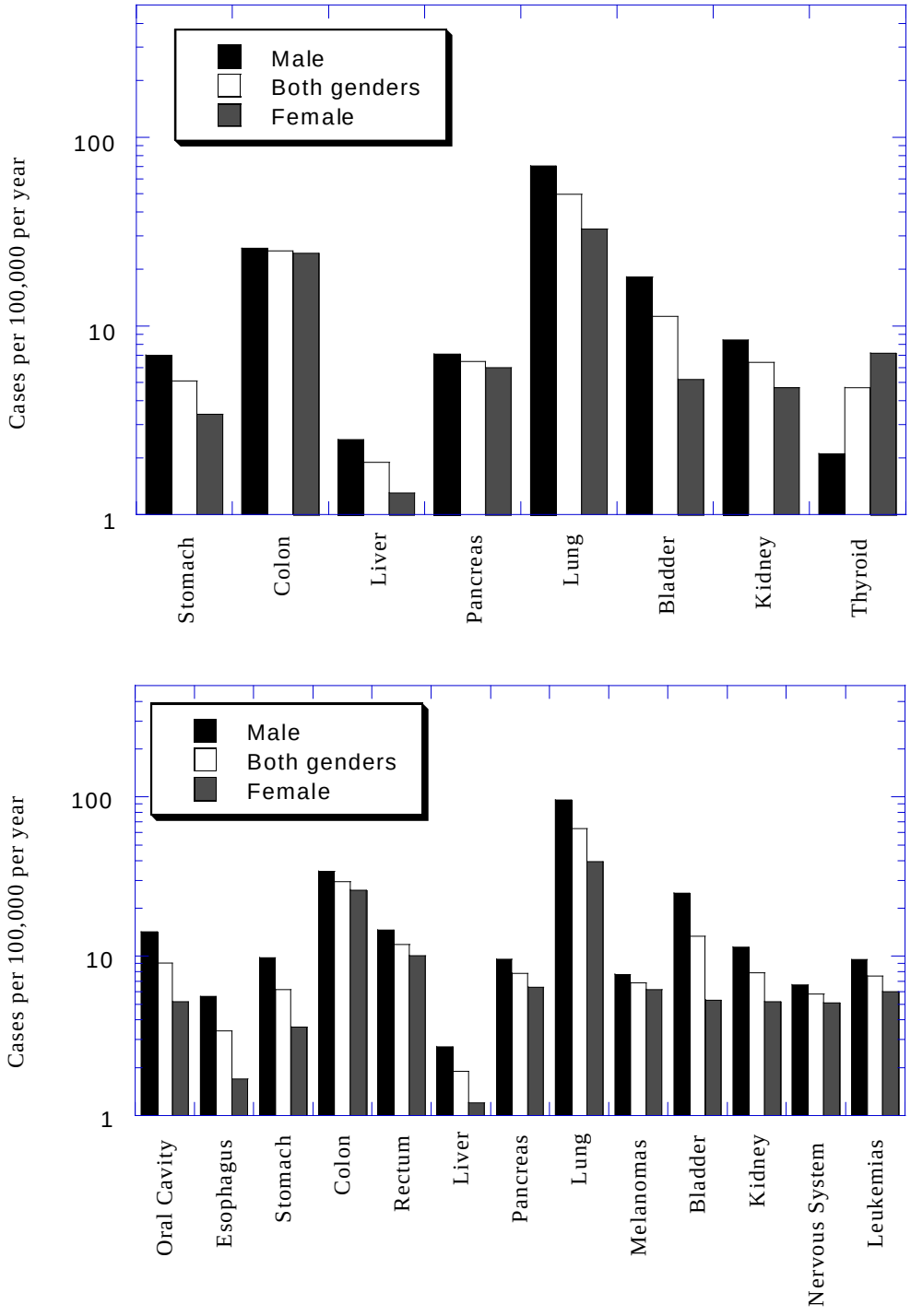

Figure 12.3 Gender differences in the age-adjusted cancer incidence rates for (a) all counties in Tennessee other than Anderson, Blount and Roane (top), and (b) the entire State of Tennessee (bottom). The rates are from observations in 1992 and are adjusted to the 1970 U.S. standard population.

(a factor[6] of 1.4 to 1.6 greater for males than for both genders), stomach (a factor of 1.4 to 1.7), liver (a factor of 1.4 to 1.6), bladder (a factor of 1.6 to 2.5), and kidneys (a factor of 1.3 to 1.5); for all of these organs the incidence rates for males are much greater than the rates for females (Figure 12.3).

To investigate the effect of a gender-specific approach, the present study used female and male background incidence rates (B) (TCRS, 1992) and the excess relative risk per unit dose ($ERR_{1\,Sv}$) for females and males (Thompson et al., 1994). Assuming the approach described in Section 12.2, the ratio (R) of the risk estimates obtained for both genders to the risk estimates obtained for females is equal to the ratio of the gender-specific parameters:

$$R = \left(\frac{B_{both\ genders}}{B_{females}} \right) \cdot \left(\frac{ERR_{1\,Sv\ both\ genders}}{ERR_{1\,Sv\ females}} \right) \qquad (12.7)$$

The analogous relationship can be written for males. Point estimates of this ratio were calculated for various organs using background incidence rates for both data sets from the Tennessee Department of Health and the excess relative risk per unit dose from Thompson et al. (1994) (Table 12.13). The ratios were estimated only for the cancer types for which the background incidence rates and the excess relative risk per unit dose are available in both data sets. A ratio larger than one means that the current approach is producing larger risk estimates than a gender-specific approach. For males, ratios larger than one are found for oral cavity, esophagus, stomach, lung and bladder, while for females, ratios are larger than one for oral cavity, stomach, colon, liver, bladder, and nervous system.

A larger risk estimate for males than the value calculated with the present approach would be produced by using the male-specific parameters for colon (10% to 20%), liver (a factor of 2), pancreas (a factor 1.3 to 1.4), and nervous system (10%) (Table 12.13). For females, the organs with a higher risk estimates would be esophagus (a factor of 3), rectum (a factor of 2), and lung (a factor of 1.25).

12.2.5.2 Influence of Age-at-Exposure

The present analysis does not explicitly consider age-dependency in risk estimation for any exposure pathways or radionuclides, other than ingestion of ^{131}I. For this radionuclide, a special treatment was

[6] The factors refer to the ratio of the incidence rates for males to that for both genders, or the ratio of the incidence rates for both genders to that for females. The range covers all values for the two data sets shown in Figure 12.3.

Table 12.13 Influence of a gender-specific approach in risk estimation for individuals exposed to radionuclides released to the Clinch River.

Ratio of the risk estimates from the current approach (both genders) to the risk estimates from the alternative approach (gender-specific)

	Males		Females	
Cancer type	**First data set[a]**	**Second data set[b]**	**First data set[a]**	**Second data set[b]**
Oral Cavity	1.2		1.1	
Esophagus	4.3		0.3	
Stomach	1.1	1.3	1.1	0.9
Colon	0.8	0.9	1.2	1.1
Rectum	--[c]		0.5	
Liver	0.5	0.5	4.6	4.3
Pancreas	0.7	0.8	2.0	1.8
Lung	1.3	1.4	0.8	0.8
Bladder	1.1	1.3	1.1	1.0
Nervous System	0.9		1.1	

[a]1992 background incidence rates for the entire State of Tennessee.
[b]1992 background incidence rates for all counties in Tennessee other than Anderson, Blount and Roane.
[c] No response to radiation was found for rectal cancer among males in the Japanese A-bomb survivors cohort (Thompson et al., 1994).

applied (Section 12.3). For all other situations, the reference individual was defined as an adult. The influence of the choice of the background incidence rates of cancer (all ages versus adults only) could be investigated by using the second set of data (Section 12.2.2.1) from the Tennessee Department of Health in Nashville (Section 12.2.2.1) (Figure 12.3). The incidence rates for adults only (ages 20 and older) are consistently larger than the rates for all ages by a factor of 1.3 to 1.4 (Figure 12.4.)

The Atomic Bomb Survivors data do not provide one single estimate of excess relative risk per unit dose for all individuals over the age of 20 or older. But, estimates were available for adults in ages 20-39, and 40 or older. To investigate the effect of both the background cancer incidence rates and the excess relative risk per unit dose on the final risk estimate, data for individuals age 40 or older were used. The investigation was similar to the one for studying the effect of gender. The ratios of the risk estimates based on the age-adjusted parameters to the risk estimates based on parameters for individuals age 40 or older were calculated as follows:

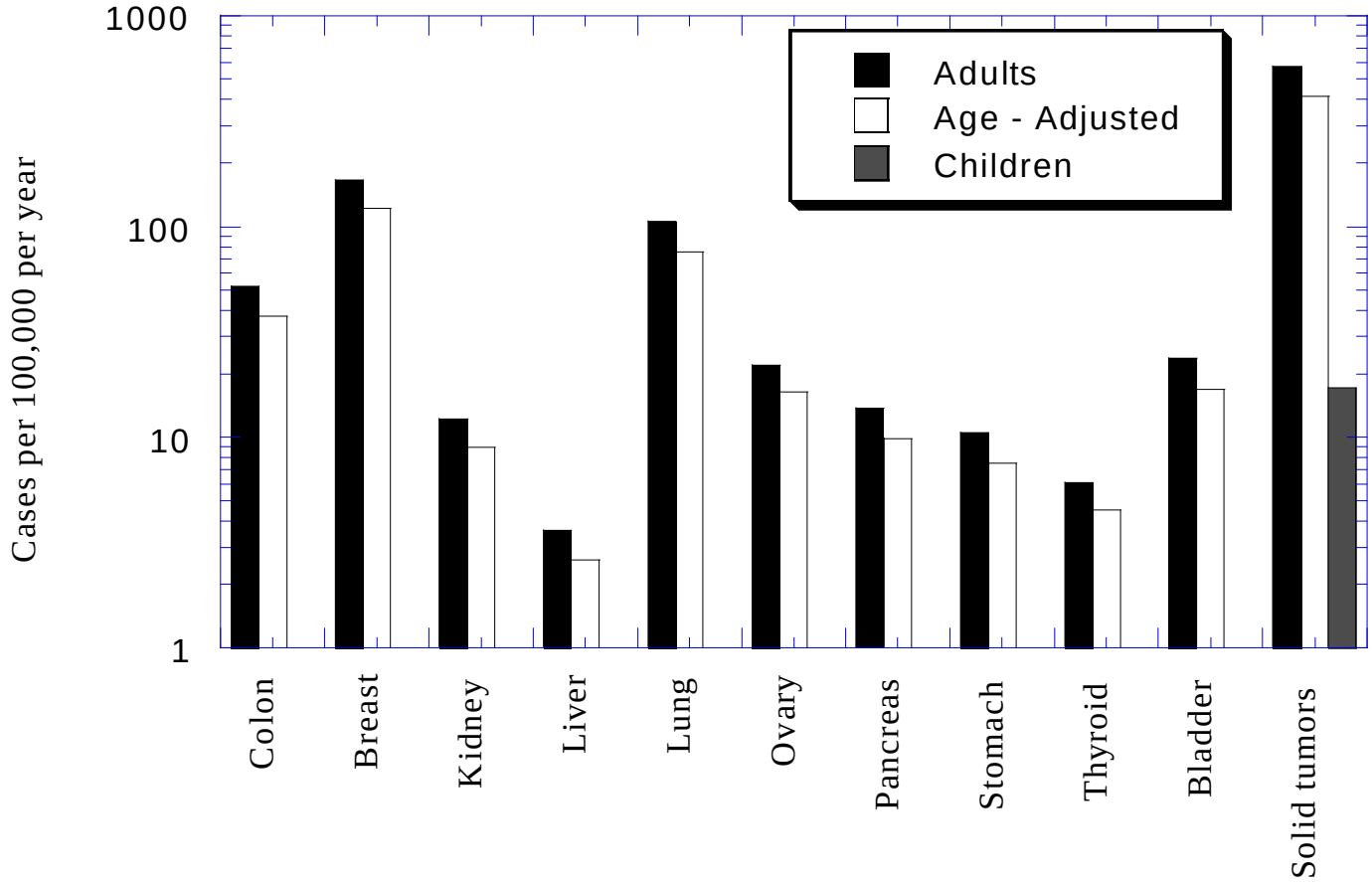

Figure 12.4 Cancer incidence rates for the State of Tennessee in 1992. The rates are adjusted to the population in Tennessee for different age groups: (a) adults 20 years of age and older, (b) individuals of all ages, and (c) children and teenagers 0 - 19 years of age. The rates are not adjusted to the standard U.S. population.

$$R' \left(\frac{B_{all\ ages}}{B_{>40\ yrs}} \right) \cdot \left(\frac{ERR_{1\ Sv\ all\ ages}}{ERR_{1\ Sv>40\ yrs}} \right) \qquad (12.8)$$

The ratios are presented in Table 12.14. If the parameters for individuals age 40 or older were used in this analysis, the risk estimates for total solid tumors would be the same for all practical purposes. Using an age-adjusted approach based on data for a population over 40, risk estimates would be higher than the ones produced in this report for the following organs: colon (a factor of 1.4), liver (a factor of 1.4), lung (a factor of 2), ovary (a factor of 3), and bladder (a factor of 2). Using an age-adjusted approach based on data for a population over 40, risk estimates would be lower than the ones produced in this report for the following organs: stomach (a factor of 1.6), pancreas (an order of magnitude), and breast (a factor of 1.7). The differences might be more pronounced for women, because women over 40 have a higher incidence of "female" related cancers.

Table 12.14 Influence of the choice of a different age group on the final risk estimates for individuals exposed to radionuclides released to the Clinch River.

Cancer Site	**Ratio of the risk estimates from the current approach (all ages) to the risk estimate from an alternative approach (individuals age > 40)**
Total Solid Tumors	1.0
Stomach	1.6
Colon	0.7
Liver	0.7
Pancreas	10.3
Lung	0.5
Breast	1.7
Ovary	0.3
Bladder	0.5

12.2.6 Total Excess Risk of Cancer Incidence

The estimation of the total excess risk of cancer incidence (all cancer types) can be very complex or even impossible, due to the lack of complete information required to determine radiation doses or risk coefficients for every organ. From the follow-up of the Japanese A-Bomb Survivors cohort (Thompson et al., 1994), a dose-response relationship for cancer incidence has been established for 22 main cancer

types (Tables 12.2). No dose-response relationship was established for the remaining cancer types, which are less common. Also, dose-response relationship is not available for cases of metastatic cancer that cannot be associated with a certain cancer site. Such cases are called "cancers with ill-defined primary site." However, a statistically significant excess risk was determined for the total number of solid tumors.

One approach to estimate the total risk is to use a risk coefficient for the excess incidence of all solid tumors and a weighted whole body dose. This dose is determined by summing the doses to individual organs weighted according to their relative importance to the total number of cancers:

$$D_w = \sum_T k_T \times D_T \qquad (12.9)$$

where:

D_w = weighted dose equivalent (Sv)
k_T = tissue weighting coefficient, and
D_T = absorbed dose equivalent in tissue T (Sv)

The tissue weighting coefficient (k_T) for an organ T measures the relative contribution of the radiogenic cancers in tissue T to the total number of radiogenic cancers. This coefficient depends on the radiosensitivity of the tissue; that is, if equal dose is delivered to all tissues, the most radiosensitive organ will make the largest contribution to the total number of cancers in the exposed population.

The “weighted” dose introduced in this study is similar in concept to the “effective” dose used by ICRP (1991). However, the ICRP “effective” dose is based on the cancer mortality data, as opposed to the “weighted” dose defined here which is based on cancer incidence. Thus, the terminology used in this report was intentionally chosen to be different from the terminology used by ICRP. That is, ICRP uses a set of “tissue weighting factors” (w_T), while this report defines a set of “tissue weighting coefficients” (k_T). Also, the "remainder" category of cancers is similar in concept in both ICRP and this study. ICRP (1991) includes in the "remainder" category the following tissues and organs: adrenals, brain, upper large intestine, small intestine, kidney, muscle, pancreas, spleen, thymus and uterus. A single weighting factor is assigned by ICRP for the remainder. In this study, the remainder contains the above mentioned cancers and any additional ill-defined cancers other than pancreatic cancer for which a specific tissue weighting coefficient was estimated.

Estimation of the doses to a tissue T is described in Section 10 for external dosimetry and in Section 11 for radionuclides taken into the body with water and food. The derivation of the tissue weighting coefficients (k_T) is presented below.

The total number of cancers in a population will be equal to the summation of individual cases of cancer found in each tissue/organ:

$$\sum_T N_T = N_{POP} \qquad (12.10)$$

where

N_T = number of solid tumors for tissue T in a given population, and
N_{POP} = total number of solid tumors in the same population.

By assuming a linear dose-response relationship (Thompson et al., 1994), the number of solid tumors originating in tissue T can be calculated as

$$N_T = D_T \times ERR_{T(1sv)} \times B_T \times P \qquad (12.11)$$

where

D_T = radiation dose to tissue T (Sv)
$ERR_{T(1sv)}$ = excess relative risk per unit dose for tissue T (Sv^{-1}),
B_T = background rate of cancer for tissue T, and
P = number of people in the population.

Likewise, the total number of solid tumors (all types) can be calculated as

$$N_{POP} = D_w \times ERR_{1sv} \times B \times P \qquad (12.12)$$

where

D_w = weighted dose equivalent (Sv) as defined earlier,
ERR_{1sv} = excess relative risk per unit dose for all solid tumors (Sv^{-1}),
B = background rate of cancer for all solid tumors, and
P = number of people in the population.

The background rate of cancer incidence for tissue T (B_T) is defined as the number of new cancers in that tissue in the unexposed population over a given period of time divided by the number of people in the unexposed population. The background rate of cancer incidence for all solid tumors (B) is defined as the total number of new solid tumors in the unexposed population over a given period of time divided by the number of people in the unexposed population.

By applying Equations 12.11 and 12.12 to Equation 12.10, we get

$$\sum_T \left(D_T \times ERR_{T(1sv)} \times B_T \times P \right) = D_w \times ERR_{1sv} \times B \times P \qquad (12.13)$$

Solving for the effective dose yields

$$D_w = \sum_T \left[\left(\frac{ERR_{T(1sv)} \times B_T}{ERR_{1sv} \times B} \right) \times D_T \right] \qquad (12.14)$$

By relating Equation 12.9 to Equation 12.14, we derive a definition for the tissue weighting factor:

$$k_T = \frac{ERR_{T(1sv)} \times B_T}{ERR_{1sv} \times B} \qquad (12.15)$$

Thompson et al. (1994) have analyzed the follow-up data for the Japanese Atomic Bomb Survivor Study (the most complete information set dealing with the relationship between the dose and risk of cancer incidence for humans). This published report estimates the excess relative risk per sievert for solid tumors for each of several tissue systems ($ERR_{T(1sv)}$). The excess relative risk per sievert for total solid tumors (ERR_{1sv}) is also estimated. Equation 12.15 uses the ratio of B_T to B, which is numerically equal to the ratio of the number of cancers originating in tissue T in the unexposed population (N_T) to the total number of solid tumors in the unexposed population (N). These numbers are given in Table 12.15.

In this report, the remainder category contains cancers having a primary site different from the listed primary sites and cancers having an ill-defined primary site. The tissue-weighting coefficient for the remainder category is calculated as one minus the sum of the other tissue coefficients.

Table 12.15 Number of reported cases of cancer occurring in the Japanese and the Tennessean populations.

Tissue or Organ System	**Unexposed Japanese population[1]**	**Tennessee population[2]**
Oral cavity and pharynx	68	515
Esophagus	101	193
Stomach	1353	368
Colon	234	1769
Rectum	172	693
Liver	302	103
Gallbladder	152	--
Pancreas	118	464
Nasal cavity	21	--
Larynx	43	--
Lung	423	3580
Skin (non-melanoma)	77	--
Female breast	240	2985
Uterus	375	852
Ovary	67	402
Prostate	79	3078
Bladder	95	799
Kidney	39	--
Renal pelvis and ureter	14	--
Nervous system	56	302
Thyroid	96	178
Remainder[3]	161	2601
Total solid tumors	4286	18882

[1] Data from Thompson et al., 1994.
[2] Data from Tennessee Cancer Reporting System (TCRS), for the entire State of Tennessee in 1992.
[3] The "remainder" category contains cancers having a primary site different from the listed primary sites and cancers having an ill-defined primary site.

Estimation of Total Risk

The excess lifetime risk of cancer incidence from all solid tumors (ELR_{ST}) is estimated as follows:

$$ELR_{ST} = D_w \times ERR_{ST(1sv)} \times B \times 70 \qquad (12.16)$$

where

D_w	=	weighted dose equivalent for all solid tumors (Sv),
$ERR_{ST(1sv)}$	=	excess relative risk per unit dose for all solid tumors (Sv^{-1}),
B	=	background rate of cancer incidence for all solid tumors (y^{-1}), and
70	=	the average lifetime of the individual (y).

Leukemia is not a solid tumor, so risk of leukemia must be added to the solid tumor risk to obtain the total risk. The excess lifetime risk of cancer incidence from leukemia (ELR_L) is estimated as follows:

$$ELR_L = D_{RBM} \times ERR_{L(1sv)} \times B_L \times 70 \qquad (12.17)$$

where

D_{RBM}	=	dose to red bone marrow (Sv),
$ERR_{L(1sv)}$	=	excess relative risk per unit dose for leukemia (Sv^{-1}),
B_L	=	background rate of cancer incidence for leukemia (y^{-1}), and
70	=	the average lifetime of the individual (y).

The total excess lifetime risk of cancer incidence (*TELR*) is estimated as follows:

$$TELR = ELR_{ST} + ELR_L \qquad (12.18)$$

Application to the U.S. Population

Tissue weighting coefficients have been calculated for a combination of both genders. However, these weighting coefficients are applicable only to the Japanese population. In order to obtain weighting coefficients which could be applied to another population (e.g., the State of Tennessee), background incidence rates for that population are needed. The ratio of cancer incidence in Tennessee to cancer incidence in Japan brings up several interesting points. Stomach cancer is much more prevalent in Japan (32% of all solid tumors) than in Tennessee (2% of all solid tumors). In contrast, lung, breast, and prostate cancers make up 51% of solid tumors in Tennessee; the same three cancers represent only 18% of solid

tumors in Japan. Therefore, it is clear that the tissue weighting coefficients derived for the Japanese population cannot be applied to doses estimated for people living in Tennessee.

To obtain weighting coefficients applicable to the Tennessee population, the weighting coefficients for the Japanese population were multiplied by the ratios of background cancer incidence (Tennessee vs. Japan).

$$k_{w(TN)} = k_{w(Japan)} \times \left(\frac{\left(B_T / B\right)_{TN}}{\left(B_T / B\right)_{Japan}} \right) \qquad (12.19)$$

The weighting coefficients for the Tennessee population were then normalized so that their sum is equal to 1.

The weighting coefficients calculated for both the Japanese and Tennessee populations are given in Table 12.16. Although not completely comparable, the weighting factors suggested by ICRP 60 (1991) are given in Table 12.16 as a reference. However, it should be noted that the ICRP values are based on cancer mortality data, while the values calculated in this report are based on cancer incidence data.

One discrepancy between the calculated factors in this report (k_T) and the values reported in ICRP 60 (w_T) is the additional weight given by the ICRP to genetic effects from irradiation of the gonads. ICRP 60 not only considers cancer mortality for doses to the gonads, but also genetic effects produced by the irradiation of the gonads.

It should be noted that the weighting factors given in ICRP 60 are composed of four components: the probability of attributable fatal cancer, the weighted probability of attributable nonfatal cancer, the weighted probability of severe hereditary effects, and weighting factor for the relative length of life lost. In addition, the background rate of cancer mortality used by ICRP is an arithmetic average of incidence of cancer mortality from six individual populations from around the world. In contrast, the weighting coefficients developed in this report for the incidence of cancer morbidity are directly applicable to the Tennessee population.

The estimation of the total cancer incidence that would occur from the intake of multiple radionuclides can be estimated using a risk coefficient for the excess incidence of all solid tumors and a weighted whole body internal dose from all radionuclides. The methodology described above was used to calculate tissue weighting coefficients which were used to sum the doses to individual organs, weighted according to their relative importance to the total number of cancers that would occur if the whole body were uniformly irradiated. Weighting coefficients directly applicable to a Tennessee population were developed.

Table 12.16 Tissue weighting coefficients (k_T) for combined gender based on data from Thompson et al. (1994), (Japan), and TCRS (1996), (Tennessee) for cancer incidence, compared with the tissue weighting factors (w_T) defined by ICRP (1991) for the derivation of the effective dose.

Cancer Site	**Japan (k_T)**	**Tennessee (k_T)**	**ICRP 60[1] (w_T)**
Oral cavity and pharynx	0.007	0.010	--
Esophagus	0.010	0.003	0.05
Stomach	0.160	0.008	0.12
Colon	0.062	0.082	0.12
Rectum	0.013	0.009	--
Liver	0.055	0.003	0.05
Gallbladder	0.007	--	--
Pancreas	0.008	0.005	--
Lung	0.149	0.220	0.12
Skin (Non-melanoma)	0.029	--	0.01[2]
Female breast	0.141	0.306	0.05
Uterus[3]	--	--	--
Ovary	0.025	0.026	0.2[4]
Prostate	0.008	0.058	--
Bladder	0.035	0.052	0.05
Nervous system	0.005	0.005	--
Thyroid	0.043	0.014	0.05
Remainder	0.242[5]	0.199[5]	0.05[6]

1 Tissue weighting factors from ICRP 60 are based on cases of weighted detriment for defining the "effective dose."
2 Since ICRP 60 tissue weighting factors are based on cancer mortality, "skin" is probably based on cases of melanoma.
3 Evidence suggests that the uterus has a negative dose-response function.
4 Value given in ICRP 60 for gonads.
5 In this report, the remainder category contains cancers having a primary site different from the listed primary sites (Table 12.2) and cancers having an ill-defined primary site.
6 ICRP 60 considers the remainder to be composed of the following tissues and organs: adrenals, brain, upper large intestine, small intestine, kidney, muscle, pancreas, spleen, thymus, and uterus.

Through the use of this method and the availability of background cancer incidence rates specific for a given location, tissue weighting coefficients can be calculated for any population.

12.3 Excess Lifetime Risk of Thyroid Cancer per Unit Dose for Children Exposed to ^{131}I

It is well established that x-ray and gamma ray irradiation of the thyroid in children under 15 years of age will result in an increased incidence of thyroid carcinomas and adenomas (NRC, 1990; Ron et al., 1995). Although fatal thyroid cancers are rare, the thyroid gland in children has one of the highest risk coefficients of any organ. Convincing evidence for increased risk occurs at just below 0.1 Gy, and linearity best describes the dose-response relationship (Ron et al., 1995). The excess relative risk (ERR) is the relative increase in the rate of cancer incidence above that expected in an unexposed population. The excess absolute risk (EAR) is the average number of cases of thyroid cancer observed above the expected amount per 10,000 person years of follow-up examination.

A pooled analysis of seven studies on thyroid cancer after childhood exposure to external radiation (Ron et al., 1995) produced an excess relative risk per Gy (ERR Gy^{-1}) of 7.7 (95% confidence interval = 2.1 to 28.7) and an excess absolute risk per 10,000 persons per year (EAR $10^{-4}\ y^{-1}$) of 4.4 (95% confidence interval = 1.9 to 10.1). The risk of thyroid cancer attributable to radiation exposure at 1 Gy was 88%. The ERR was highest for females exposed to a single treatment before the age of 5 and lowest for males over the age of 10 who received more than a single treatment, with the ratios of relative risks for these two groups being about a factor of 14.

The pooled analysis by Ron et al. (1995) included 5 cohort studies and two case-control studies totaling approximately 120,000 persons (58,000 exposed subjects and 61,000 nonexposed subjects), in which nearly 700 thyroid cancers were seen during 3,000,000 person years of follow-up examinations.

Unlike the evidence for x-rays and gamma rays, the effectiveness of ^{131}I in producing thyroid cancer is a subject associated with considerable uncertainty (Laird, 1987; Van Middlesworth, 1989; Shore, 1992). Epidemiological studies of about 3500 Utah school children exposed to ^{131}I from atmospheric weapons testing at the Nevada Test Site have produced a significant dose response for all categories of thyroid neoplasms with an ERR per Gy of about 7.0 (90% confidence interval. = 0.7 to 33), but the number of carcinomas (8) and adenomas (11) in this cohort is small (Stevens et al., 1992, Kerber et. al., 1993).

The most convincing evidence, however, is still emerging from reports and preliminary results from the follow-up of children in the aftermath of the Chernobyl accident of 1986. Since 1990, about 1,000 cases of childhood thyroid cancers have been identified within Belarus, Russia and Ukraine, and it is apparent that this observed increase in thyroid cancer is the result of exposure to ^{131}I (Karaoglou et al., 1996). A preliminary dose-response relationship for children under the age of 15 (EAR of about 4 cases per 10,000 individuals per Gy per year) was reported by Dr. V. Beral (1996) at the 1996 annual meeting of the National Council on Radiation Protection and Measurements for exposures to Chernobyl ^{131}I in Belarus, Russia and Ukraine. From analysis of the data as published by Sinclair (1996), an excess relative risk per

unit dose of 41 ERR Gy^{-1} was derived. This result is confirmed by Jacob et al. (1998), who found an excess absolute risk for children under the age of 15 of 2.4 (95% confidence interval of 1.4 to 3.8) per 10^4 person-year Gy for the three countries. The differences between the derived excess absolute risk are quite small: from 1.0 per 10^4 person-year Gy in Zhytomyr county (Ukraine) to 3.5 per 10^4 person-year Gy in Gomel county (Belarus). The excess relative risk from the same set of studies (Jacob et al., 1998) was 46 ERR Gy^{-1}, with values ranging from 21 ERR Gy^{-1} in Zhytomyr county to 85 ERR Gy^{-1} in the city of Kiev (Ukraine). The large values of the excess relative risk per unit dose from exposure to ^{131}I released from Chernobyl can be explained by the short follow-up period of the exposed cohort (1991 to 1995) and by the young age of the people in the cohort. At this young age, the natural incidence of thyroid cancer is low. As the cohort ages, the background incidence of thyroid cancer increases, while the radiosensitivity of the thyroid decreases. Therefore, it is expected that the lifetime excess relative risk per unit dose from the Chernobyl cohort will be much lower (see also Section 12.2.2.5). In conclusion, the values derived from the epidemiological studies of the Chernobyl cohort cannot be directly used to estimate the lifetime risk of thyroid cancer, but they are useful in bringing some light to the issue of ^{131}I effectiveness in producing thyroid cancer (Section 12.3.1.4).

Similar effects for x-rays and ^{131}I were observed for the induction of thyroid carcinomas in prepubescent female Long-Evans rats (Lee et al., 1982); these effects were observed regardless of the dose rate at which the animals were exposed. In these experiments, an excess lifetime risk of 1.9×10^{-2} per Gy from exposure to ^{131}I was found. The average lifetime of the Long-Evans rats is about 24 months. Lee et al. (1982) estimated that children exposed to x-ray and gamma rays at age 10 should have an excess lifetime risk of 2.1×10^{-2} per Gy (assuming a human lifetime of 73 years).

12.3.1 Methodology Used for Estimation of the ^{131}I Risk Factors

Based on the epidemiological evidence (Ron et al., 1995), the risk of acquiring thyroid cancer from exposure to a radiation dose *D* can be expressed by a linear relative risk model. Values of the excess lifetime risk per unit dose from an ^{131}I exposure at age *i* (the risk factors) are obtained as follows:

$$RF_i = B_i \cdot b_i = B_i \cdot \left(b \cdot g \cdot \in_i / d \right) \qquad (12.20)$$

where

RF_i = the excess lifetime risk of thyroid cancer per unit dose from exposure at age "*i*" per unit absorbed dose [Gy^{-1}];

B_i = the background lifetime risk of thyroid cancer for a person exposed at age "*i*" [unitless] (see Section 12.3.1.1);

β_i = the age- and gender-dependent excess relative risk of thyroid cancer per unit dose for exposure to ^{131}I [Gy^{-1}];

β = the excess relative risk (ERR) per unit dose for females 0-4 years of age [Gy^{-1}];

g = a modifying factor accounting for differences between genders [unitless];

ε_i = a modifying factor accounting for the age at time of exposure [unitless];

δ = a modifying factor accounting for the relative effectiveness of ^{131}I in producing thyroid cancer as compared with x-rays and gamma rays [unitless].

Each of the parameters is described below.

12.3.1.1 The Background Lifetime Risk of Thyroid Cancer (B_i)

An individual of a given age "*i*" has a background risk of thyroid cancer for the duration of his or her remaining life. This background risk is called "lifetime background risk" (B_i); it depends on the age ("*i*") and on the life expectancy of the individual. In this study, the exposed individual is an average individual having an average lifetime of 70 years.

By definition, the individual had no thyroid cancer at the age in question. If the individual already had thyroid cancer, the disease would not have been induced by exposure to ^{131}I, and thus he or she would not be a subject of this study. If the individual is exposed at age "*i*," the lifetime background incidence of thyroid cancer can be estimated as shown in Equation 12.21. The derivation of this relationship is presented in Appendix 12A.

$$B_i = \sum_{k=i}^{lifetime} \left(R_{0,k} \cdot \Delta t\right) \qquad (12.21)$$

where

$R_{0,k}$ = age-specific thyroid cancer incidence rate [yr^{-1}] (Figure 12.5)

Δt = 1 year

The age-specific thyroid cancer incidence rates for Tennessee were provided by the Tennessee Department of Health in Nashville (Bashor, 1996). These data are gender specific, and they are separated into three groups: (a) incidence rates for all Tennessee counties, (b) incidence rates for Anderson, Roane, Loudon and Knox counties, (c) incidence rates for all counties in Tennessee other than Anderson, Roane, Loudon and Knox. The latter data were used for estimation of the background risk of thyroid cancer both because they are considered to be specific for Tennessee areas and because they represent people that most probably had not been exposed to ^{131}I released from the Oak Ridge Reservation. The calculated

background lifetime risk was found to be almost constant for exposures at ages less than 14 years (Figure 12.6). This effect can be explained by the low incidence rate in childhood as compared with the rates in adulthood.

For exposures at ages less than 14 years, the background lifetime risk for females for all counties in Tennessee other than Anderson, Roane, Loudon and Knox is $B_i = 3.6 \times 10^{-3}$ for an average lifetime of 70 years. The main source of uncertainty in the estimation of the lifetime background risk of thyroid cancer (B_i) is introduced by using an average lifetime of 70 years. A given individual may live longer than or less than 70 years. For a lifetime of 85 years or more, the calculated background lifetime risk was 5.2×10^{-3} (Figure 12.6). The ratio between the background for 85 years and the background for 70 years was calculated ($5.2 \times 10^{-3} / 3.6 \times 10^{-3} = 1.4$). This ratio was used as an uncertainty factor and applied to the background for an average lifetime of 70 years. The calculated range for the background lifetime risk of thyroid cancer for females in Tennessee exposed under the age of 4 is 2.6×10^{-3} to 5.2×10^{-3}. A loguniform distribution was assigned to this parameter. A similar rationale was applied for all ages and for both genders, and the uncertainty to the background incidence rate was assigned appropriately (Table 12.17).

Table 12.17 Background lifetime risk of cancer for all counties in the State of Tennessee other than Anderson, Roane, Loudon and Knox, obtained assuming a life span of 70 years.

Age at Exposure [yr]	Background Lifetime Risk for a Lifespan of 70 years	Uncertainty Factor[a]	lower limit	upper limit
females				
0-4	3.6×10^{-3}	1.4	2.6×10^{-3}	5.2×10^{-3}
5-9	3.6×10^{-3}	1.4	2.6×10^{-3}	5.0×10^{-3}
10-14	3.6×10^{-3}	1.4	2.6×10^{-3}	5.0×10^{-3}
15-19	3.5×10^{-3}	1.5	2.3×10^{-3}	5.2×10^{-3}
20-29	3.2×10^{-3}	1.5	2.1×10^{-3}	4.8×10^{-3}
30-39	2.4×10^{-3}	1.7	1.4×10^{-3}	4.0×10^{-3}
males				
0-4	1.4×10^{-3}	1.6	8.8×10^{-4}	2.3×10^{-3}
5-9	1.4×10^{-3}	1.6	8.8×10^{-4}	2.3×10^{-3}
10-14	1.4×10^{-3}	1.6	8.7×10^{-4}	2.2×10^{-3}
15-19	1.39×10^{-3}	1.6	8.7×10^{-4}	2.2×10^{-3}
20-29	1.27×10^{-3}	1.7	7.5×10^{-4}	2.2×10^{-3}
30-39	1.07×10^{-3}	1.8	5.9×10^{-4}	1.9×10^{-3}

[a]The uncertainty factor is obtained as a ratio between the background lifetime risks for a lifespan of more than 85 years and a lifespan of 70 years. The uncertainty is described as a log-uniform distribution between the lower and upper limits obtained by applying the uncertainty factor to the central value calculated for a lifespan of 70 years. Numbers are rounded to two significant digits.

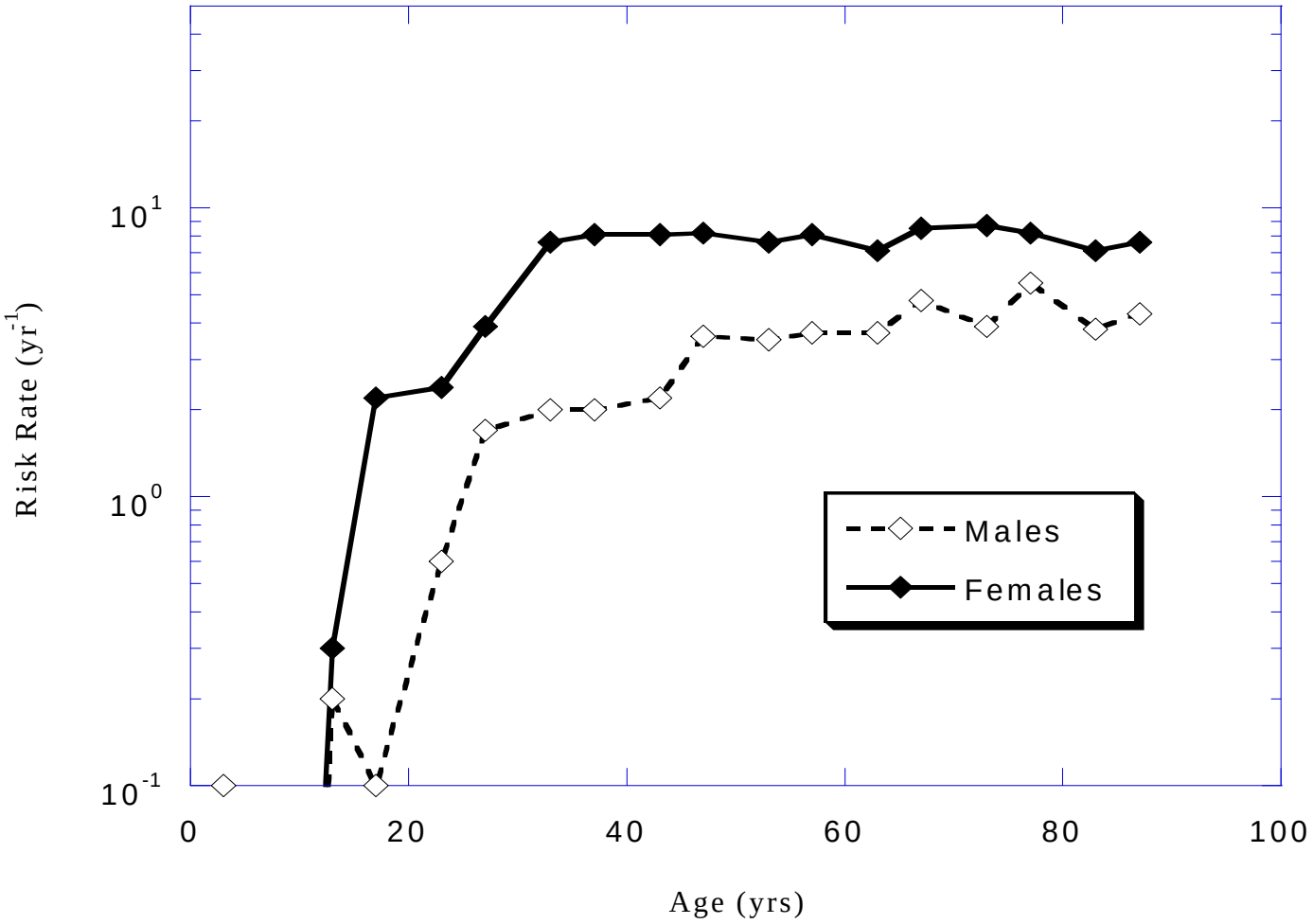

Figure 12.5 Age-specific thyroid cancer rates for males and females (Tennessee data).

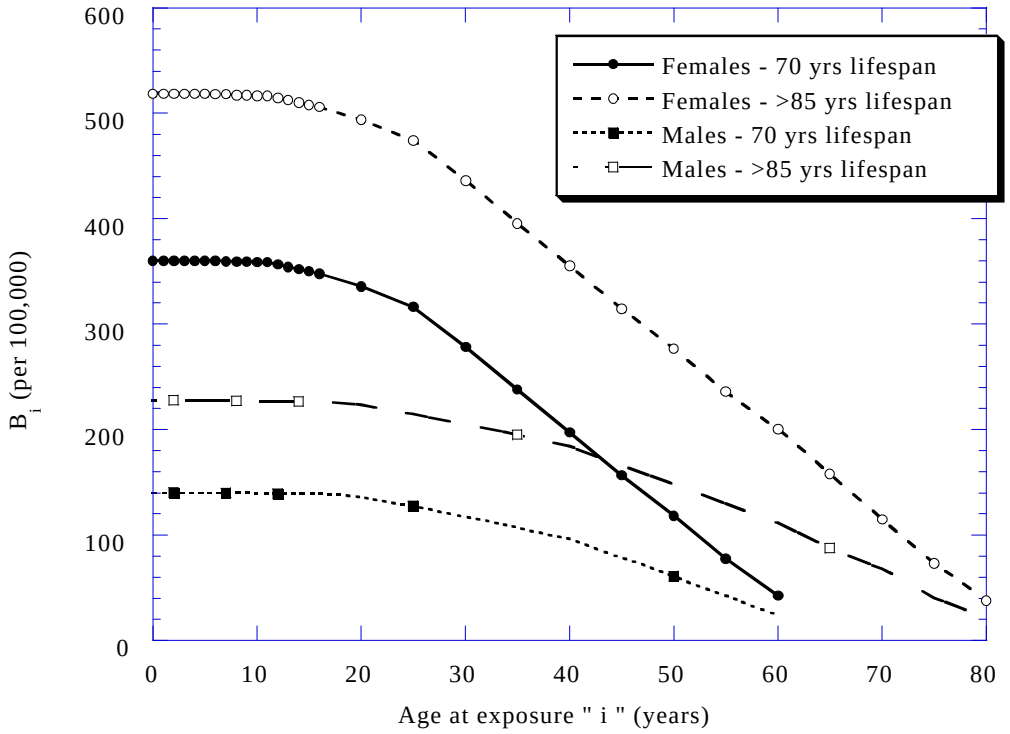

Figure 12.6 Background lifetime risks of thyroid cancer based on statistics for all counties in the State of Tennessee other than Anderson, Roane, Loudon, and Knox counties.

12.3.1.2 The Excess Relative Risk of Thyroid Cancer per Unit Dose (β)

The most comprehensive source of information about the radiation-induced thyroid cancer is the analysis of thyroid cancer after childhood exposure to external radiation by Ron et al. (1995). This study found an excess relative risk per Gy (ERR Gy^{-1}) of 7.7 (95% confidence interval = 2.1 to 28.7) for childhood exposure (<15 years) of children of both genders. The most at-risk individuals are found to be females exposed to a dose of external radiation of 0.1 Gy or more before age of 5.

To indicate the relative differences between the risk for other age and gender groups and the risk for females under age of 5, Ron et al. (1995) report a set of age-at-first-exposure and gender modifying factors (Table 12.18) (these factors are also discussed in Section 12.3.1.3). However, the study does not report an excess relative risk for females under age of 5 (β). This quantity (β) has to be derived starting from the reported excess relative risk for childhood exposures (7.7 ERR Gy^{-1}):

$$\beta = 7.7 \cdot \frac{n}{\left(a_{<1} \cdot \gamma_{<1} + a_{1\&4} \cdot \gamma_{1\&4} + a_{5\&9} \cdot \gamma_{5\&9} + a_{10\&14} \cdot \gamma_{10\&14}\right)} \cdot \frac{n}{\left(g_{male} \cdot \gamma_{male} + g_{female} \cdot \gamma_{female}\right)} \qquad (12.22)$$

In Eq. 12.22, the quantities denoted by $\tilde{a}$ are the age and gender modifying factors reported by Ron et al. (1995) (Table 12.18). The quantities denoted by a and g are the number of cases for the different age-groups and genders, respectively. In the cohort analyzed by Ron et al. (1995), 202 cases of thyroid cancer were found in males and 271 in females (n = 473). The numbers of cases ($a_{age\text{-}group}$) in each age group are as follows: 64 children exposed as newborns (< 1 year old), 251 children exposed between 1 and 4 years of age, 111 children exposed between 5 and 9 years old, and 47 children exposed between 10 and 14 years of age (Table 12.18).

The "best estimate" of 7.7 (ERR Gy^{-1}) from Ron et al. (1995) is used as the geometric mean of a lognormal distribution, and the 95% confidence interval of 2.1 to 28.7 is used to obtain a geometric standard deviation of 1.95. The excess relative risk for females exposed under the age of 5 was found to be 12.2 (ERR Gy^{-1}), with a 95% confidence interval of 3.3 to 44.5.

Adolescents older than 15 years of age and adults are less susceptible to radiation-induced thyroid cancer (Ron, 1996; Ron et al., 1995). The excess relative risk per unit dose for people in this group was estimated at 0.4 (ERR Gy^{-1}) with a 95% confidence interval of -0.1 to 1.2 (Ron et al., 1995). A lognormal distribution with a mean of 0.4 and a standard deviation of 0.41 was assigned to describe the uncertainty in the ERR per unit dose. The standard deviation was obtained from the reported 95% confidence interval.

Table 12.18 Pooled Analysis of Cohort Studies of Persons Exposed before Age 15 Years: Modifiers of Excess Relative Risk per Gy (ERR Gy^{-1}) (Ron et al., 1995).

Variable	Categories	Number of Cases	Modifying Factor ($\tilde{a}$)	P value
Gender	Male	202	0.5	0.07
	Female	271	1.0	
Age at first exposure (years)	<1	64	1.0	0.004
	1-4	251	1.0	
	5-9	111	0.5	
	10-14	47	0.2	
	15-19		1.0[a]	
	20-39		0.6[a]	
	40		0.0[b]	
Number of treatments	1	373	1.0	0.18
	2+	63	0.7	

[a]Data on exposure as adults are based on the Japanese Atomic Bomb Survivors cohort analyzed by Ron et al. (1995).

[b]No dose-response relationship for thyroid cancer can be found for people exposed after the age of 40 (Thompson et al. 1994).

Finally, the analysis of the Japanese Atomic Bomb Survivors shows a negative dose-response relationship for adults exposed after age 40 (Thompson et al., 1994). Thus, no risk associated with ^{131}I exposure is assumed in this report for individuals over the age of 40 at the time of exposure.

12.3.1.3 Age and Gender Modifying Factors (g, ε_i)

Ron et al. (1995) also present a number of factors (Table 12.18) that can result in a net increase or decrease in the excess relative risk per unit dose (ERR Gy^{-1}). These modifying factors were applied to the excess relative risk per unit dose for females under age of 5 at the time of exposure.

To account for the dependency on the age at first exposure, a modifying factor of 1 is used for the age group of 0 to 4 years; for the age groups of 5 to 9 years and 10 to 14 years, modifying factors are 0.5 and 0.2, respectively. The effect of age at time of exposure on the annual risk of thyroid cancer is shown in Figure 12.7.

For adolescents, a specific excess relative risk per unit dose was used (see Section 12.3.1.2). Thus, the modifying factor related to the age at exposure for this age-group is 1. For adults under the age of 40, a modifying factor of 0.6 was used, as reported by Ron et al. (1995) from an analysis of the Japanese Atomic Bomb Survivors. For adults exposed after 40 years of age, no positive dose-response relationship is found (Thompson et al., 1994).

Gender is another important factor influencing the estimation of the radiation-induced risk of thyroid cancer. Ron et al. (1995) found a tendency for females to be twice as sensitive as males in acquiring thyroid cancer from exposure to external radiation (although this finding was only of marginal statistical significance (P = 0.7). Nevertheless, in our risk calculation a modifying factor of 0.5 is assigned for males consistent with the central tendency reported by Ron et al. (1995) (Table 12.18).

12.3.1.4 Iodine-131 Effectiveness Factor (δ)

National and international organizations have stated that an absorbed dose (Gy) of ^{131}I should be at least a factor of three less effective than a similar absorbed dose of shorter-lived isotopes of iodine or of x-rays and gamma rays (ICRP, 1991; UNSCEAR, 1994; NCRP, 1985; USEPA, 1994). These statements appear to be supported by the publications of Shore (1992) and Hall et al. (1996). The problem is that most of the epidemiological data on ^{131}I are for diagnostic and therapeutic doses applied to adolescents and adults. In adults, especially those exposed after 40 years of age, the effect of exposure to ^{131}I is similar to that for x-rays and gamma rays: no conclusive dose-response relationship has been shown.

At the 1996 annual meeting of the National Council on Radiation Protection and Measurements (NCRP), it was concluded that data were insufficient to support past claims that a given dose of ^{131}I is less effective than the same dose of x-rays and gamma rays (Ron, 1996). Similar concerns have been expressed by Laird (1987) and Van Middlesworth (1989). The few epidemiological data that are available on the effects of ^{131}I doses to the thyroid for individuals exposed in childhood show a dose response that is quite similar to those reported for x-rays and gamma rays.

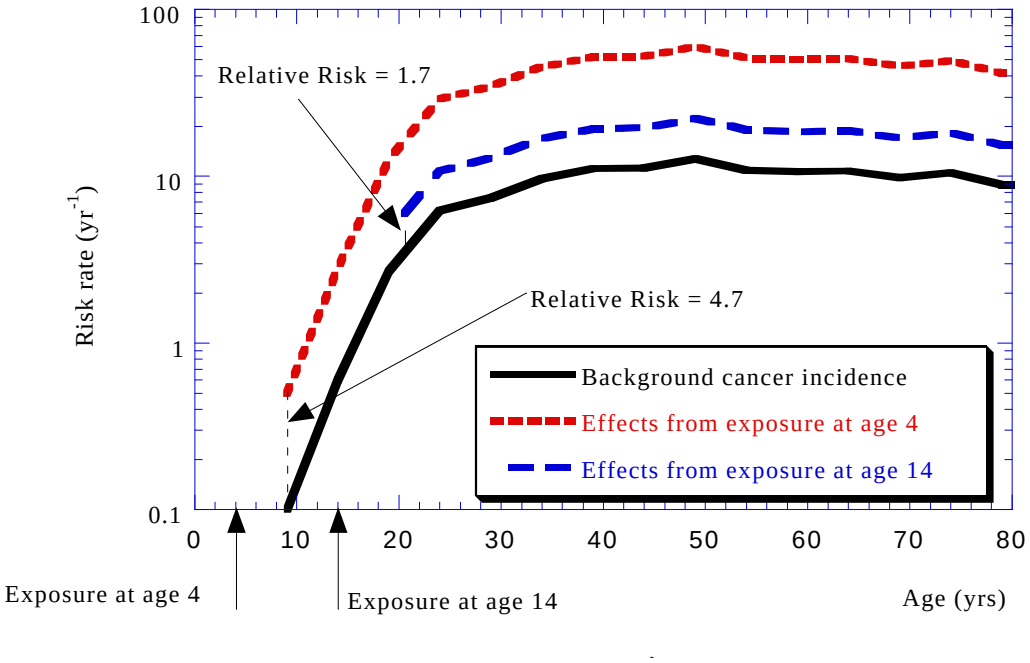

Figure 12.7 Effect of age at time of exposure on the annual risk of thyroid cancer for a single (external) dose of 30 rad.

After review of the literature (Jacob et al. 1998; Ron, 1996; Hall et al., 1996; Ron et al., 1995; Shore, 1992; Van Middlesworth, 1989; Laird, 1987; Lee et al., 1982; ICRP, 1991; UNSCEAR, 1994; NCRP, 1985; USEPA, 1994; Johnson and Myers, 1983) and after discussions with outside experts, we believe that the state of knowledge for this parameter is currently described by a range from 1.0 to 5.0. The weight of evidence now seems to support values much closer to 1.0 than to 5.0.

To capture the uncertainty associated with this parameter we use two approaches. The first assumes no difference in the effectiveness between external radiation and ^{131}I. The support for this assumption comes from recent observations of the incidence of thyroid cancer in children exposed to ^{131}I in the aftermath of the Chernobyl accident (Beral, 1996; Jacob et al. 1998), the dose response observed in Utah residents exposed in childhood to ^{131}I from weapons testing at the Nevada Test Site (Stevens et al., 1992, Kerber et al., 1993), and the animal study performed by Lee et al. (1982).

The second approach treats the effectiveness of ^{131}I as an uncertain variable with subjective weight given to values of 1.0, 1.5, 2.0, 3.0, and 5.0 (Figure 12.8). In this approach, a subjective weight of 35% is assigned to the value of 1.0, given the strong evidence that there may not be a difference between exposure to external radiation and exposure to ^{131}I. A subjective weight of 40% is assigned to the value of 1.5, given a tendency to the data from fractionated exposures to external radiation to show a slightly lower effect over time than if the dose is delivered as a single unit exposure (Ron et al., 1995). We note, however, that this effect was not statistically significant.

Subjective weights of 15%, 7%, and 3% are assigned to the values of 1.5, 3.0, and 5.0, respectively. The possibility of these lower values is given much lower subjective weight due to the absence of direct supportive evidence, but acknowledging the recommendations of national and international committees for a reduction of the effectiveness of radiation-induced cancer when exposure involves a low dose rate, as is the case with exposures to prolonged releases of ^{131}I to the air and water from the X-10 facility (UNSCEAR, 1994; ICRP, 1991; NCRP, 1985; 1997; USEPA, 1994).

The possibility of a low ^{131}I effectiveness is also included because there is no epidemiological evidence demonstrating an increase in thyroid cancer resulting from long-term, low dose-rate exposures to children spanning a period of several years to a decade or more. One major reason for this, of course, is that few opportunities have existed whereby such long-term exposure situations have been available to epidemiologic investigation.

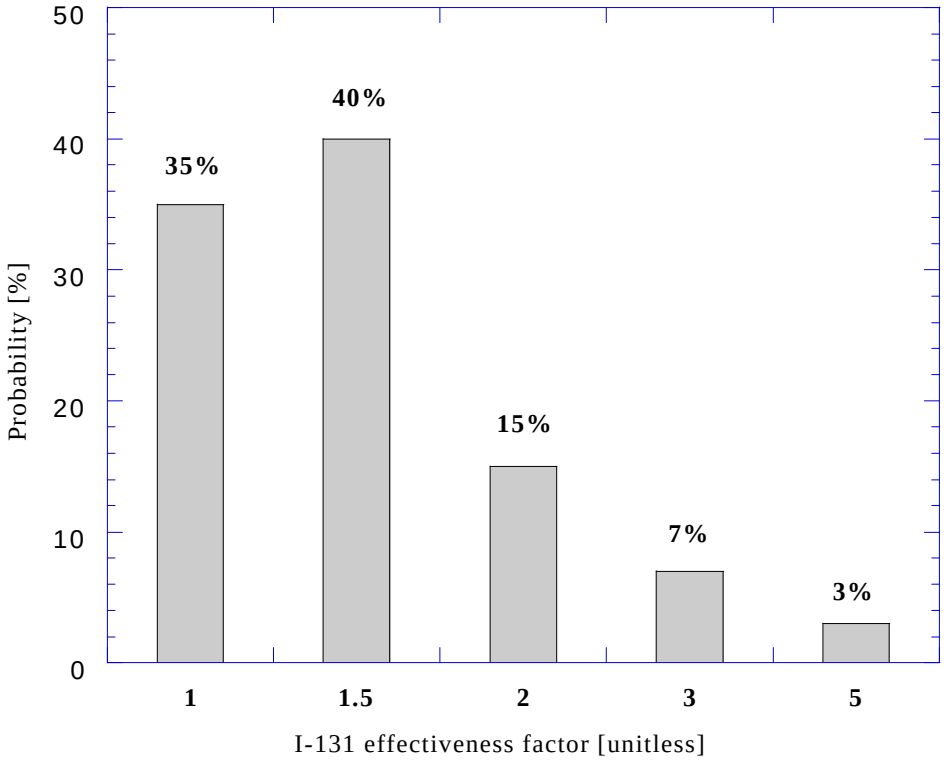

Figure 12.8 Discrete probability distribution function that describes the uncertainty in the ^{131}I effectiveness factor.

Possible contradictions to this statement are the Utah study (Kerber et al., 1993) in which children were exposed to ^{131}I in Nevada Test Site fallout over a period of years, and the outcome of cohorts of children exposed to multiple functionalities of x-ray therapy, extending over a period of several years (Ron et al., 1995). Additional information may become available in 1998 with the completion of the Hanford Thyroid Disease Study; this is a case-control study of children in eastern Washington exposed to ^{131}I releases to the atmosphere over a period of years from fuel reprocessing operations at Hanford.

12.3.2 Estimates of ^{131}I Risk Factors

The 95% confidence interval for the excess lifetime risk of thyroid cancer per unit dose for females exposed under the age of 5 was found to range between 6.3×10^{-3} Gy^{-1} and 1.3×10^{-1} Gy^{-1}, with a best estimate of 3.0×10^{-2} Gy^{-1} (Table 12.19), or 3.0×10^{-4} rad^{-1} (Figure 12.9). The risk for exposure at older ages is still high during childhood, but it decreases substantially (about two orders of magnitude) for exposure as adults. Males have a lower risk of acquiring thyroid cancer by about a factor of 2 (Figure 12.9).

Table 12.19 Estimates of the excess lifetime risk per unit dose (risk factors) [Gy^{-1}].

Age at Exposure	**Excess Lifetime Risk per unit dose (Risk Factor) [Gy^{-1}]**		
	2.5%-tile	median	97%-tile
Females			
0-4	6.3×10^{-3}	3.0×10^{-2}	1.3×10^{-1}
5-9	2.9×10^{-3}	1.6×10^{-2}	6.9×10^{-2}
10-14	1.2×10^{-3}	6.3×10^{-3}	2.6×10^{-2}
15-19	9.7×10^{-5}	6.5×10^{-4}	4.6×10^{-3}
20-29	5.1×10^{-5}	3.7×10^{-4}	2.2×10^{-3}
30-39	3.5×10^{-5}	2.8×10^{-4}	1.6×10^{-3}
Males			
0-4	1.2×10^{-3}	5.8×10^{-3}	2.6×10^{-2}
5-9	5.7×10^{-4}	2.9×10^{-3}	1.3×10^{-2}
10-14	2.5×10^{-4}	1.2×10^{-3}	4.8×10^{-3}
15-19	7.5×10^{-6}	5.9×10^{-5}	3.6×10^{-4}
20-29	3.8×10^{-6}	2.9×10^{-5}	1.8×10^{-4}
30-39	3.3×10^{-6}	2.6×10^{-5}	1.5×10^{-4}

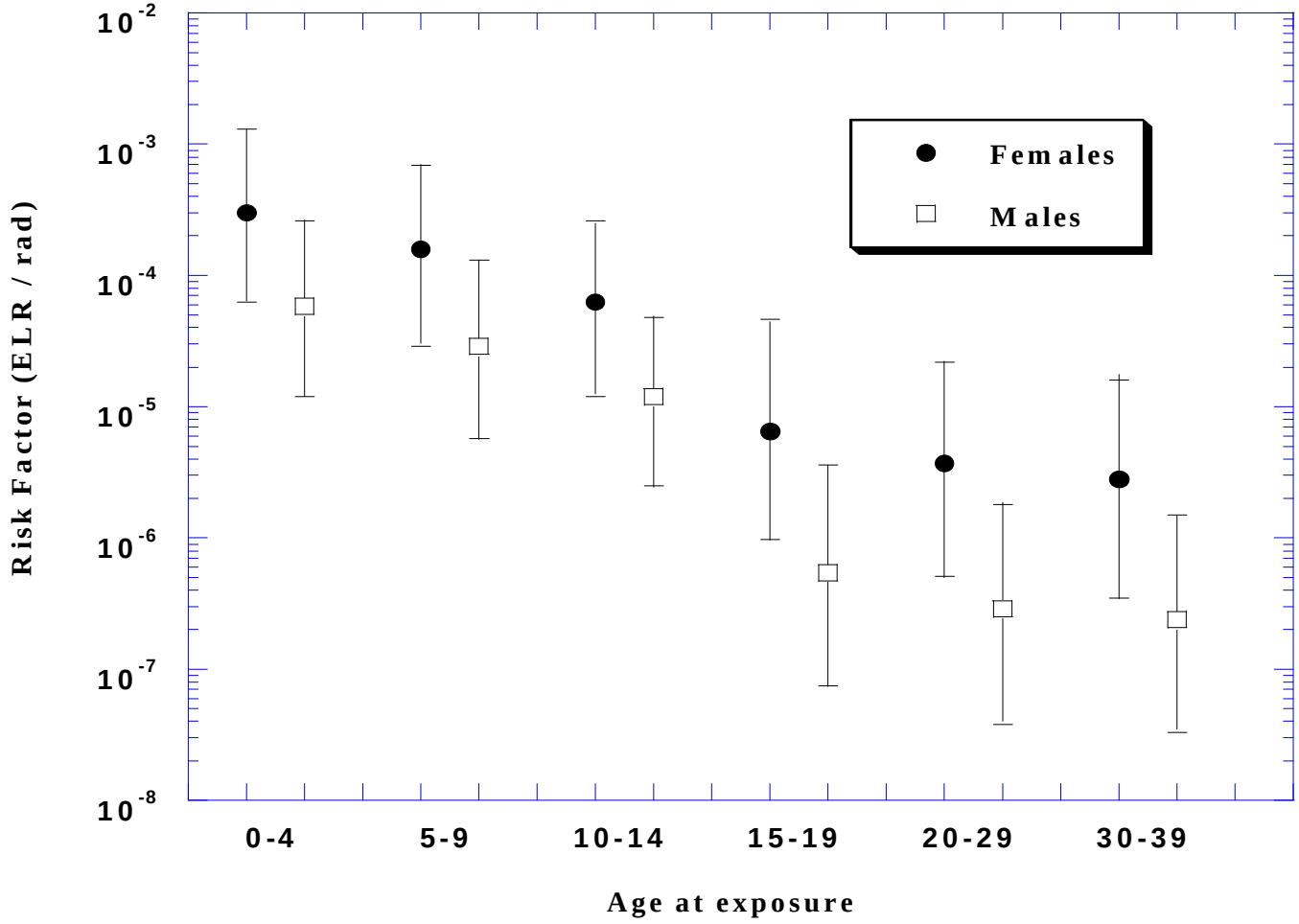

Figure 12.9 Excess Lifetime Risk of thyroid cancer per 0.01 Gy (1 rad); Estimated Risk Factors.

The risk factors described here were obtained by expressing the differences in thyroid cancer induction between internal exposure to ^{131}I and external exposure to x-rays and gamma rays as an uncertain effectiveness factor having a distribution as described in Section 12.3.1.4. The average value of the ^{131}I effectiveness factor from this distribution is 1.6. The extreme situation, as indicated by the recent studies of thyroid cancer incidence in children exposed to ^{131}I released during the Chernobyl accident (Jacob et al., 1998; Beral, 1996), is that ^{131}I is as effective as x-rays and gamma rays in inducing thyroid cancer; that is, the ^{131}I effectiveness factor is 1. If this were the real case, the average risk estimates should be larger by a factor of 1.6. A second estimation of the risk factors was performed assuming that the effectiveness factor is1; the comparison between the two sets of results is shown in Figure 12.10. Since an effectiveness factor equal to 1 is not yet totally confirmed, and in the presence of remaining high uncertainty associated with the preliminary results from investigations in Russia, Belarus, and Ukraine, we base the risk estimates in this report in the uncertainty on the effectiveness factor as defined in Section 12.3.1.4.

12.3.3 Sensitivity Analysis

To identify the most important contributors to the uncertainty, a sensitivity analysis was performed for the excess lifetime risk per unit dose for females of all age groups (Table 12.20). These results apply for males also, because no uncertainty is associated with the gender modifying factor. The uncertainty in the risk factors is dominated by the large uncertainty in the excess relative risk per unit dose for all age groups. The ^{131}I effectiveness factor is the second contributor to the uncertainty, while the uncertainty introduced by the lifetime background risk of thyroid cancer is minimal.

Table 12.20 Sensitivity analysis for the excess lifetime risk per unit dose for females exposed at various ages

Parameter	Contribution to Uncertainty					
	age 0-4	**age 5-10**	**age 10-14**	**age 15-19**	**age 20-29**	**age 30-39**
Excess Relative Risk per Gy	75%	75%	77%	82%	82%	77%
^{131}I effectiveness factor	18%	18%	16%	13%	15%	14%
Background cancer incidence	8%	8%	8%	4%	3%	9%

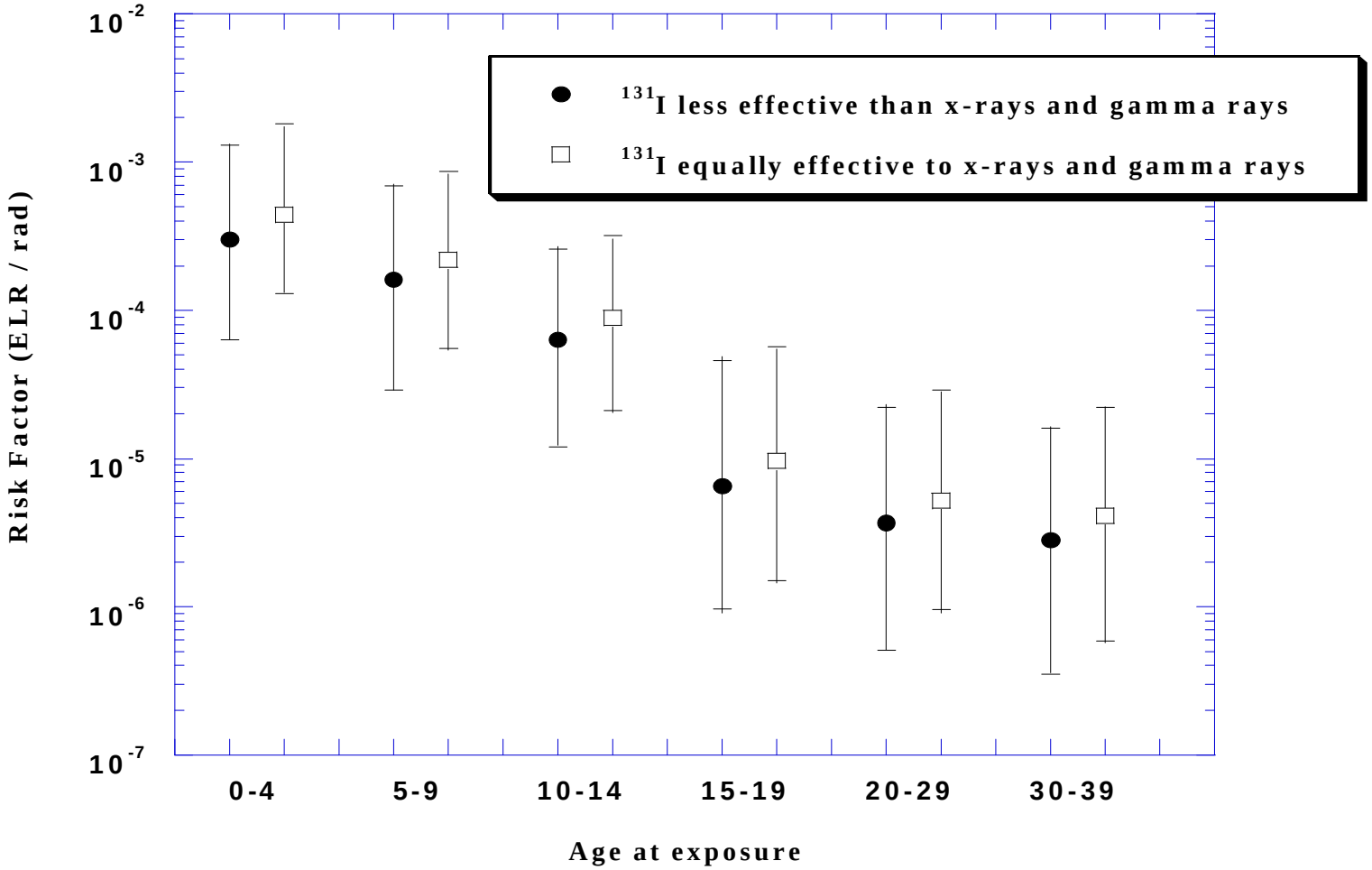

Figure 12.10 Comparison between different assumptions about ^{131}I effectiveness in the estimation of the Excess Lifetime Risk of thyroid cancer per 0.01 Gy (1 rad) for females.

12.4 Summary

The health effects on members of the public of the radioactivity released from the Oak Ridge Reservation can be measured in terms of risk of radiation-induced cancer incidence. This study reports the excess lifetime risk of cancer incidence, which is defined as the risk of developing radiation-induced cancer, in excess of the background risk of the incidence of cancer, at some point in life after the first exposure to radiation.

This section summarizes the available data on the relationship between radiation dose and cancer incidence. The main source of information is the epidemiological study performed for the Japanese Survivors of the Atomic Bombs detonated in Hiroshima and Nagasaki (Thompson et al., 1994). For radiation-induced thyroid cancer, the outcome of a pooled analysis of seven epidemiological studies (Ron et al., 1995) was used.

Based on the available data, *risk factors* (representing the excess lifetime risk of cancer incidence from a unit dose of radiation) were developed for 18 different types of cancer. The risk factors are age- and gender-adjusted; that is, they are designed to be applied to a reference individual in the population. In contrast, female children are most susceptible to acquiring thyroid cancer when the thyroid gland is exposed to radiation. Thus, age- and gender-specific risk factors were developed for thyroid cancer. The most radiosensitive organs are the female breast, red bone marrow (for induction of leukemia), lung, and colon (Table 12.11). For thyroid cancer, the most sensitive people are girls exposed under the age of 4 (Table 12.19).

The risk factors were customized for the Tennessee population; that is, they are based on the background risk of cancer incidence in different areas of the State of Tennessee (Table 12.3). To the extent possible, the background incidence of cancer was chosen for populations not exposed to the radioactivity released from the Oak Ridge Reservation (Table 12.A.1 and 12.A.2).

A sensitivity analysis (12.2.4) was performed to identify the most important contributors to the uncertainty in the risk factors for selected organs. Such a contributor is the dose-rate effectiveness factor, which accounts for differences between the response to a high dose rate (for A-bomb survivors) and exposure situations involving doses delivered over a longer period of time at a much lower rate. Another important source of uncertainty is the lack of knowledge about which model (absolute versus relative) is correct for transferring the information from A-bomb survivors to a U.S. population.

In addition, this section presents details on the methods and philosophy employed for risk estimation for a given cancer type. Also, a special method was designed for estimating the total risk of cancer incidence. Finally, a significant improvement to the present study would be an age- and gender-specific analysis of the health risk for specific individuals exposed to the radionuclides released to the Clinch River.

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13.0 RESULTS OF THE DOSE AND RISK CALCULATIONS FOR HISTORICAL RELEASES OF RADIONUCLIDES TO THE CLINCH RIVER

This section contains results obtained by combining the following components:

- estimated concentrations of radionuclides in Clinch River water and shoreline sediments (Sections 5 and 6);
- the transfer of radionuclide contamination to aquatic and terrestrial biota (Sections 8 and 9);
- internal and external exposure parameters for pathways of concern (Section 7);
- the radiation doses per unit exposure (Sections 4, 10 and 11); and
- the risk of cancer incidence per unit dose (Section 12).

The approach and methods described in Section 4 were used to estimate internal and external exposures and the resulting doses and risks, based on the estimated radionuclide concentrations in water and shoreline sediment for each year and each location of concern.

Reference individuals for each of four locations along the Clinch River were described in terms of the exposure pathways that could have occurred at that location (Section 13.1). Estimates of doses (Section 13.2) and risks (Section 13.3) to these reference individuals were based on the assumption of continued exposure at the indicated location from 1944 to 1991. In addition, because the X-10 operations and releases varied over time, selected analyses of doses and risks were also performed for shorter periods of exposure (Sections 13.4 and 13.5). The reference individuals for which dose and risk estimates were made are described according to the pathways that could have occurred for each of four sections of the river.

Risk of cancer incidence is the overall endpoint of the study; therefore, most of the discussion is devoted to the description of results in terms of risk. The results from this analysis are compared to results obtained using the recently published Federal Guidance 13, which provides government consensus of regulatory values of risk per unit intake (Section 13.6), and the most important sources of uncertainty in the estimates of dose and risk are identified through a series of sensitivity analyses (Section 13.7).

The results of this study are discussed in terms of their potential implications for individuals consuming fish from Watts Bar Reservoir (Section 13.8). In addition, these results are compared with findings from earlier investigations on the Clinch River and other sites where radionuclides have been released to river systems (Section 13.9). The possible contribution to the estimated doses and risks of radionuclides from weapons fallout is also discussed (Section 13.10).

The primary radionuclides of interest released from the X-10 facility into the Clinch River via White Oak Creek were identified in the initial screening analysis as ^{137}Cs, ^{90}Sr, ^{60}Co, ^{106}Ru, ^{144}Ce, ^{131}I, ^{95}Zr, and ^{95}Nb (Section 3). Of these radionuclides, ^{137}Cs, ^{60}Co, ^{106}Ru, ^{90}Sr, and ^{131}I were evaluated for their contribution to internal exposure pathways, while ^{137}Cs, ^{60}Co, ^{106}Ru, ^{90}Sr, ^{144}Ce, ^{95}Zr, and ^{95}Nb were evaluated for their contribution to the external exposure pathway. Although more than 90% of the activity released into the Clinch River was from tritium (^{3}H), tritium was not included in the detailed analysis. Tritium does not accumulate in sediments or in biota; it is diluted substantially on its release into the Clinch River, and upon reaching Watts Bar Reservoir (which takes about five days), it is further diluted (USDOE, 1996). In addition, tritium is a weak beta emitter, and its dose per unit activity is very low. In the initial screening analysis, which was based on conservative, upper-bound release estimates and exposure assumptions (Section 3), the total estimated risk from tritium from all pathways and for all years combined was below a level of 1×10^{-5}, more than 10 times lower than the decision criterion established by the ORHASP for screening assessments.

13.1 Summary of Reference Individuals and Exposure Pathways

Of seven potential exposure pathways evaluated in the initial screening analysis (Section 3), five were determined to have occurred historically (Section 7): ingestion of Clinch River fish, ingestion of drinking water from the Clinch River, external exposure from shoreline sediments, ingestion of milk from cows having access to river water, and ingestion of beef from cattle having access to river water. Reference individuals, for whom dose and risk calculations have been made, were described for each of four locations along the Clinch River: near Jones Island (CRM 20.5); near the K-25 site or Grassy Creek (CRM 14); near the Kingston Steam Plant (CRM 3.5); and near the city of Kingston (CRM 0). For each location, the reference individuals were specified in terms of the exposure pathways that occurred near that site (Tables 13.1 and 13.2). Reference individuals were described as adults (males and females) for all situations except internal exposure to ^{131}I. Dose and risk factors for ^{131}I are age-specific, with only children below age 15 having a significant risk of radiation-induced thyroid cancer from ingestion of ^{131}I. Dose and risk factors for the other radionuclides in this analysis are not age-specific. Therefore, the reference individuals for the drinking water and milk ingestion pathways included both children (for ^{131}I) and adults (for ^{137}Cs, ^{90}Sr, ^{106}Ru, and ^{60}Co).

Table 13.1 Summary of exposure pathways by location.

Clinch River Reach	Exposure Pathways	Exposure Period[a]
Jones Island area, CRM 20.5 (CRM 21.0-CRM 17.0)	Fish ingestion	1944-1991
	External exposure	1963-1991[b]
	Ingestion of meat and milk	1963-1991[b]
K-25/Grassy Creek areas, CRM 14 (CRM 17.0 to CRM 5.0)	Drinking water	1944-1991
	Fish ingestion	1944-1991
	External exposure	1944-1991
	Ingestion of meat and milk	1944-1991
Kingston Steam Plant, CRM 3.5 (CRM 5.0 to CRM 2.0)	Drinking water	1954-1989[c]
	Fish ingestion	1944-1991
	External exposure	1944-1991
	Ingestion of meat and milk	1944-1991
City of Kingston, CRM 0 (CRM 2.0 to CRM 0)	Drinking water	1955-1991[d]
	Fish ingestion	1944-1991
	External exposure	1944-1991
	Ingestion of meat and milk	1944-1991

[a] Unless otherwise indicated, the reference individual was assumed to have been exposed for the entire study period, 1944-1991.

[b] Public exposure to contaminated water or sediment near Jones Island occurred only on the far bank (south bank); negligible contamination occurred on that bank prior to operation of Melton Hill Dam (beginning in 1963) because the contaminated plume was not mixed until further downstream.

[c] The Kingston Steam Plant obtained drinking water from an intake on the Clinch River only from 1954-1989.

[d] The City of Kingston began to obtain drinking water from an intake on the Tennessee River (near the confluence of the Clinch River) in 1955.

Table 13.2 Summary of reference individuals by exposure pathway.

Exposure Pathway	Reference Individuals
Drinking water	Adult visitors to K-25
	Adult visitors to the Kingston Steam Plant
	Adults and children living, working, or in school in the city of Kingston
Fish ingestion	Adults consuming fish from the Clinch River
External exposure	Adults using the shoreline for recreational purposes
Ingestion of meat and milk	Adults consuming meat from farm-raised cattle
	Adults and children consuming milk from a backyard cow

Consumption of fish from any section of the river could have occurred, so the fish ingestion pathway was included at all sites. Reference individuals were specified in terms of the amount of fish consumed (Section 7):

- Category I, 1 to 2.5 meals per week
- Category II, 0.25-1.3 meals per week (1 meal per month to 1.3 meals per week)
- Category III, 0.04-0.33 meals per week (1 meal per 6 months to 1 meal per 3 weeks)

Meal size was defined as 0.10 to 0.30 kg per meal for males and 0.08 to 0.25 kg per meal for females. Thus in terms of total fish consumption per year, the categories can be summarized as follows: Category I, 7.1 to 33 kg y^{-1} for males and 5.7 to 27 kg y^{-1} for females; Category II, 2.2 to 16 kg y^{-1} for males and 1.8 to 14 kg y^{-1} for females; and Category III, 0.39 to 4.3 kg y^{-1} for males and 0.32 to 3.6 kg y^{-1} for females[*]. These values represent the 95% subjective confidence intervals of the subjective probability distributions for intake obtained by propagation (using Monte Carlo techniques) of the distributions for number of meals and amount of fish per meal; the extreme values (e.g., 1 meal per week $\times$ 0.10 kg per meal = 5.2 kg y^{-1} and 2.5 meals per week $\times$ 0.30 kg per meal = 39 kg y^{-1}) are outside the 95% subjective confidence interval. For each category, 20-100% of the fish consumed was assumed to have come from the Clinch River near the stated location (Section 7), and 80-90% of the radioactivity in the fish was assumed to remain after processing.

Exposure parameters for other pathways are described in detail in Section 7. For example, reference adults consumed an average of 0.8-2.4 L of drinking water per day (95% subjective confidence interval, 310-860 L y^{-1}); the fraction of drinking water assumed to come from the Clinch River varied by location and by type of use (i.e., residential, occupational, or both). The amount of time (days per year and hours per day) spent by a reference individual on the shoreline varied by location (95% subjective confidence intervals: 75-430 h y^{-1} at CRM 20.5, 85-440 h y^{-1} at CRM 14, and 130-490 h y^{-1} at CRM 3.5 or CRM 0).

13.2 Estimation of Organ-Specific Radiation Doses

The methods used to estimate organ-specific doses from radionuclides in the Clinch River are described in Sections 4, 10, and 11. The doses to the reference individuals from each of the five major exposure pathways are summarized in this section.

[*] These categories correspond to the following ranges (95% subjective confidence intervals) in pounds: Category I, 16-72 lb y^{-1} for males and 13-60 lb y^{-1} for females; Category II, 4.8-36 lb y^{-1} for males and 3.9-30 lb y^{-1} for females; Category III, 0.85-9.4 lb y^{-1} for males and 0.70-7.9 lb y^{-1} for females.

13.2.1 Ingestion of Fish

The highest organ doses estimated for any pathway in this study were for ingestion of fish from CRM 20.5 by Category I and II consumers. For ingestion of fish from other locations (all consumption levels) or by Category III consumers (all locations), doses ranged from slightly greater (a factor of 2-3) than to substantially less (about an order of magnitude) than doses from external exposure (Section 13.2.2). The 95% subjective confidence intervals for the organ doses received by Category I consumers near CRM 20.5 range from about 0.02 to 8 cSv (0.02 to 8 rem), with central values varying from 0.3 to 0.8 cSv for males and from 0.2 to 0.6 cSv for females (shown for selected organs in Table 13.3; a complete list of organ doses by gender, location, and consumption rate is provided in Appendix 13A). Doses to Category II consumers are about a factor of 2 less than for Category I at any given location, and for Category III about a factor of 4 less than for Category II (Table 13.3; Appendix 13A). The differences in doses between males and females or between categories of consumption reflect the differences in the consumption rates defined for each category and gender (Sections 7 and 13.1). Doses from fish obtained near CRM 20.5 are about a factor of 6 higher than for CRM 14 for similar reference individuals; doses for CRM 14 are about a factor of 1.5 higher than for CRM 3.5 and CRM 0, which are about the same. Differences in doses with location reflect differences in the estimated radionuclide concentrations in water and fish by location (Sections 6 and 8).

The highest values of organ doses were obtained for the bone and red bone marrow, and the lowest for the breast, brain, ovaries, and skin, although the greatest differences in organ doses (central values) were only a factor of 2-3 for males and 3-4 for females. The similarity of doses among organs probably reflects the importance of ^{137}Cs to the organ doses from fish ingestion, as ^{137}Cs is accumulated in the flesh of fish and then rather evenly distributed throughout the body of an individual consuming the fish. The higher doses to bone reflect the additional contribution of ^{90}Sr from the fish.

Table 13.3 Selected organ-specific doses (cSv[a]) for consumption of fish from the Clinch River near Jones Island or Kingston[b].

	Jones Island (CRM 20.5)			Kingston (CRM 0)		
Organ	**Category I**	**Category II**	**Category III**	**Category I**	**Category II**	**Category III**
Males						
Bone	0.067-7.4[c] (0.81)[d]	0.023-3.2 (0.33)	0.0046-0.78 (0.081)	0.011-0.58 (0.096)	0.0050-0.26 (0.041)	0.00086-0.066 (0.0097)
Lower large intestine	0.053-5.2 (0.57)	0.022-2.0 (0.23)	0.0043-0.49 (0.057)	0.010-0.40 (0.064)	0.0036-0.18 (0.029)	0.00069-0.045 (0.0062)
Red bone marrow	0.052-4.8 (0.60)	0.018-2.0 (0.24)	0.0043-0.49 (0.055)	0.0095-0.40 (0.065)	0.0029-0.18 (0.027)	0.00068-0.047 (0.0065)
Skin	0.027-2.9 (0.31)	0.0088-1.1 (0.13)	0.0018-0.28 (0.031)	0.0054-0.24 (0.035)	0.0017-0.097 (0.016)	0.00032-0.024 (0.0036)
Females						
Bone	0.058-7.9 (0.60)	0.019-2.7 (0.28)	0.0038-0.81 (0.059)	0.011-0.54 (0.074)	0.0041-0.21 (0.034)	0.00091-0.053 (0.0079)
Breast	0.024-3.6 (0.24)	0.0089-0.96 (0.11)	0.0018-0.40 (0.025)	0.0049-0.19 (0.030)	0.0015-0.091 (0.013)	0.00032-0.025 (0.0028)
Lower large intestine	0.046-4.5 (0.43)	0.018-1.7 (0.19)	0.0039-0.52 (0.043)	0.0091-0.29 (0.054)	0.0029-0.15 (0.023)	0.00060-0.038 (0.0050)
Red bone marrow	0.039-4.8 (0.42)	0.015-1.6 (0.20)	0.0036-0.61 (0.043)	0.0095-0.30 (0.050)	0.0024-0.15 (0.022)	0.00071-0.041 (0.0055)
Skin	0.022-2.8 (0.23)	0.0072-0.87 (0.11)	0.0018-0.40 (0.025)	0.0044-0.19 (0.028)	0.0014-0.080 (0.013)	0.00031-0.024 (0.0027)

Footnotes to Table 13.3

[a] 1 cSv = 1 rem.

[b] Consumption rates are as follows: Category I, 1-2.5 meals per week, or 7.1–33 kg y^{-1} for males and 5.7–27 kg y^{-1} for females.
Category II, 0.25–1.3 meals per week, or 2.2–16 kg y^{-1} for males and 1.8–14 kg y^{-1} for females.
Category III, 0.04–0.33 meals per week, or 0.39–4.3 kg y^{-1} for males and 0.32–3.6 kg y^{-1} for females.
For all categories, it was assumed that 20 to 100% of the fish was contaminated, and 80 to 90% of the radioactivity in the fish was retained after processing (Sections 7, 13.1).

[c] 95% subjective confidence interval

[d] central value (50th percentile)

13.2.2 External Exposure to Shoreline Sediment

For a representative adult at the K-25/Grassy Creek area (CRM 14), organ-specific doses ranged from about 0.003 to 0.2 cSv, with central values varying from 0.015 to 0.047 cSv (Table 13.4; a complete list of organ-specific doses by location is given in Appendix 13A). The highest doses from external exposure were to bone, skin, and thyroid, although the doses (central values) varied among organs by only a factor of about 3; the doses were dominated by gamma-emitting radionuclides. Estimated doses were lower for CRM 20.5 than for CRM 14, about the same at CRM 3.5, and slightly higher at CRM 0 (Table 13.4, Appendix 13A); this pattern reflects the fact that contamination of sediment on the public bank at CRM 20.5 was negligible prior to 1963 before dam operation caused mixing of the plume near CRM 20.5 (Section 7), as well as the greater radionuclide concentrations in sediment at the downstream locations than at CRM 20.5 (Section 6) and the different amount of time spent along the shoreline assumed for each of the locations (Sections 7 and 13.1). For a reference individual at CRM 14, organ-specific doses from external exposure were lower than those from ingestion of fish by a factor of 1.1-3.5 for Category I consumers (Figure 13.1). For Category II consumers, doses from external exposure ranged from a factor of 1.5 less to a factor of 2.2 greater than the corresponding doses from fish ingestion. For Category III consumers, doses from external exposure were a factor of 3-10 higher than those from ingestion of fish. Thus, for individuals with low or moderate consumption of fish but who used the Clinch River shoreline, the largest organ-specific doses from radionuclide contaminants would have been from external exposure.

13.2.3 Ingestion of Drinking Water

Estimated organ-specific doses to adults from the ingestion of filtered, treated Clinch River water as drinking water ranged from 0.00025 to 0.55 cSv at CRM 14 (central values, 0.0014-0.11 cSv), from 0.00014 to 0.24 cSv (central values, 0.00098-0.058 cSv) at CRM 3.5, from 0.0000096 to 0.039 cSv (central values, 0.00011-0.0062 cSv) for people who lived and worked in Kingston, and from 0.0000087 to 0.030 cSv (central values, 0.000083-0.0051 cSv) for people who lived or worked, but not both, in Kingston (Table 13.5; Appendix 13A). These doses reflect both the differences in water concentrations with location (Section 6) and the assumptions made about residency and water usage for each location (Section 7). Drinking water at CRM 0 (the City of Kingston) was contaminated only during periods of backflow of the Clinch River into the Tennessee (Section 7); thus doses estimated for drinking water at CRM 0 are about an order of magnitude lower than those for drinking water at CRM 14 or 3.5. The dose estimates (central values) varied across organs by factors of 50-80 for a given location. The highest doses were to the bone, red bone marrow, and upper and lower large intestine, reflecting the importance of ^{90}Sr, which is concentrated in the bone, and ^{106}Ru, which largely stays in the gastrointestinal tract, to the doses received. Doses to most other non-gastrointestinal organs were similar for any given location, reflecting the relatively even distribution of ^{137}Cs within the body.

Table 13.4 Selected organ-specific doses (cSv[a]) to adults from external exposure to Clinch River shoreline sediment, by location[b].

Organ	Jones Island (CRM 20.5)	K-25/Grassy Creek (CRM 14)	Kingston Steam Plant (CRM 3.5)	City of Kingston (CRM 0)
Bone	0.0026-0.059[c] (0.012)[d]	0.0082-0.22 (0.047)	0.0080-0.22 (0.045)	0.0089-0.24 (0.050)
Breast (female)	0.0018-0.039 (0.0090)	0.0070-0.15 (0.034)	0.0071-0.14 (0.036)	0.0076-0.16 (0.037)
Lower large intestine	0.0013-0.030 (0.0071)	0.0056-0.14 (0.027)	0.0058-0.13 (0.027)	0.0065-0.13 (0.029)
Red bone marrow	0.0016-0.038 (0.0077)	0.0052-0.13 (0.029)	0.0061-0.13 (0.030)	0.0065-0.15 (0.032)
Skin	0.0022-0.039 (0.010)	0.0088-0.16 (0.040)	0.0098-0.16 (0.044)	0.011-0.19 (0.047)

[a] 1 cSv = 1 rem
[b] 95% subjective confidence intervals for exposure time: 75 – 430 h y^{-1} at CRM 20.5, 85 – 440 h y^{-1} at CRM 14, and 130 – 490 h y^{-1} at CRM 3.5 and CRM 0
[c] 95% subjective confidence interval
[d] central value (50th percentile)

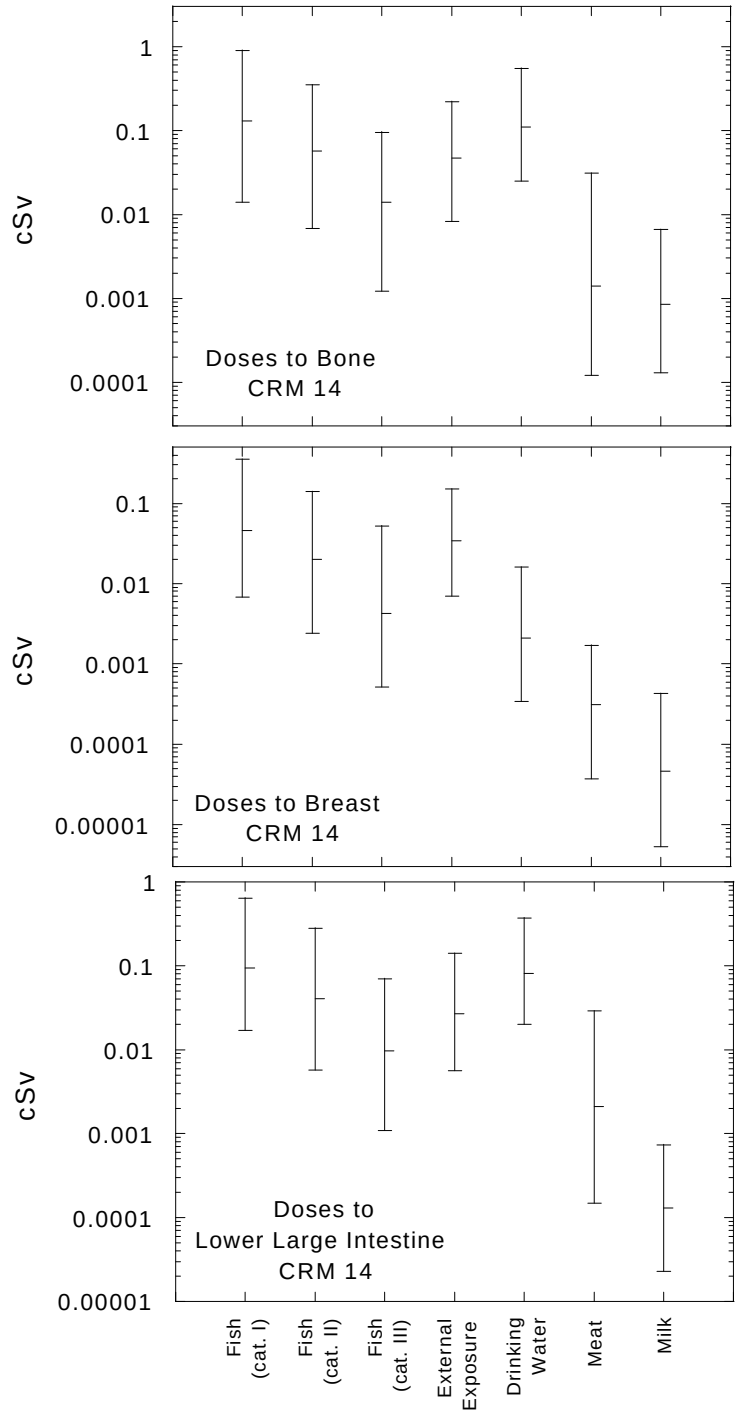

Figure 13.1 Comparison by pathway of doses to selected organs for reference individuals at CRM 14. The reference individuals for doses to bone and lower large intestine are adult males; for doses to breast, the reference individuals are adult females. For fish consumption, categories are as follows: Category I, 1 to 2.5 meals per week; Category II, 0.25 to 1.3 meals per week; Category III, 0.04 to 0.33 meals per week (Section 13.1). The vertical lines indicate the 95% subjective confidence intervals on the estimated doses; the central values (50^{th} percentiles) are also indicated.

Table 13.5 Selected organ-specific doses (cSv[a]) to adults at CRM 14 from ingestion of drinking water, meat, or milk.

Organ	Ingestion of drinking water[b]	Ingestion of meat[c]	Ingestion of milk[d]
Bone	0.025-0.55[e] (0.11)[f]	0.00012-0.031 (0.0014)	0.00013-0.0066 (0.00084)
Breast (female)	0.00034-0.016 (0.0021)	0.000037-0.0017 (0.00031)	0.0000053-0.00043 (0.000046)
Lower large intestine	0.020-0.37 (0.081)	0.00015-0.029 (0.0021)	0.000023-0.00074 (0.00013)
Red bone marrow	0.012-0.19 (0.046)	0.00010-0.017 (0.00081)	0.000065-0.0027 (0.00042)
Skin	0.00036-0.014 (0.0024)	0.000042-0.0018 (0.00031)	0.0000055-0.00042 (0.000048)

[a] 1 cSv = 1 rem.
[b] Based on ingestion of 20 to 60% of filtered, treated water from the Clinch River (Section 7).
[c] Based on ingestion of 30 to 80% of meat from cattle that had access to the Clinch River (Section 7).
[d] Based on ingestion of 63 to 100% of milk from cows that had access to the Clinch River (Section 7).
[e] 95% subjective confidence interval
[f] central value (50th percentile)

At CRM 14, doses from drinking water to bone, bone marrow, and upper and lower large intestine were greater than doses to the same organs from external exposure (Figure 13.1); for most other organs, doses from drinking water were less than those from external exposure. At CRM 0, doses from external exposure were greater than those from drinking water for all organs; at CRM 3.5, a few doses from drinking water (bone, upper and lower large intestines) were similar to or greater than those from external exposure. Doses from fish ingestion for Category I consumers were higher than those from drinking water at all locations. However, for Categories II and III at CRM 14 and 3.5, doses to bone, bone marrow, and upper and lower large intestine were higher from drinking water; at CRM 0, doses from fish ingestion (Categories II and III) were higher for all organs. Thus, for a few specific organs (bone, bone marrow, upper and lower large intestine), ingestion of drinking water at CRM 14 or 3.5 (even after treatment and filtration) could have produced higher doses than either fish consumption (low or moderate consumption rate) or external exposure (Figure 13.1). The drinking water scenario at CRM 14 is applicable to anyone (including workers) who consumed approximately 20 to 60% of their water from the K–25 Water Intake System. While the number of people consuming this amount of drinking water over the entire exposure period would be small the possibility cannot be ignored. Drinking water at CRM 0 (the City of Kingston) was contaminated only during periods of backflow of the Clinch River into the Tennessee (Section 7); thus doses estimated for drinking water at CRM 0 are substantially lower than those for drinking water at CRM 14 or 3.5, and also lower than doses estimated for external exposure or any level of fish consumption at that location.

13.2.4 Ingestion of Meat and Milk

Organ-specific doses from ingestion of meat or milk were highest at CRM 14 (Table 13.5; Appendix 13A). The lowest doses from these pathways occurred at CRM 20.5. Cattle at CRM 20.5 would have been exposed to contaminated water only after 1963; prior to the operation of Melton Hill Dam, mixing of the water did not occur until further downstream and thus the public (south) bank received negligible contamination. At CRM 14, estimated doses from ingestion of meat ranged from 0.000025 to 0.031 cSv (central values, 0.00021-0.0021 cSv), and from ingestion of milk, from 0.0000046 to 0.0066 cSv (central values, 0.000038-0.00084 cSv). For specific organs, the doses from ingestion of meat or milk are approximately 1 to 3 orders of magnitude below those from ingestion of drinking water (Figure 13.1). The highest doses from either meat or milk were to the bone, bone marrow, and upper and lower large intestine; thyroid doses from milk were slightly larger than those to the upper large intestine. Although the overall doses are small, these observations reflect the relative importance to human doses of ^{90}Sr (milk and beef) and ^{106}Ru (beef) in the water consumed by cows.

13.2.5 Estimates of Thyroid Dose for a Female Child Exposed from the Drinking Water and Milk Ingestion Pathways

Only the drinking water and milk ingestion pathways were considered for ^{131}I in Clinch River water. The reference individual was a female child (0 to 14 years of age). Females under the age

of 15 at the time of exposure are considered to be at the highest risk of radiogenic thyroid cancer later in life. Ingestion of milk by a female child was considered for two locations, K-25/Grassy Creek (CRM 14) and the Kingston Steam Plant area (CRM 3.5). Dose estimates at Kingston (CRM 0), where exposures also occurred, are about the same as those for CRM 3.5 (the same assumptions were used and approximately the same radionuclide concentrations in water existed, so a separate calculation was not made). The drinking water pathway was considered only for a child raised in Kingston (CRM 0), as this was the only location where residential drinking water could have been contaminated. Doses from milk ingestion were calculated for the fifteen-year period 1946-1960, when ^{131}I concentrations in Clinch River water were highest. Exposure via drinking water could not have occurred before 1955, so doses from drinking water and from the water and milk pathways combined were calculated for the fifteen-year period 1955-1969.

Estimated thyroid doses to a female child are presented in Table 13.6. The highest dose occurs for a female child ingesting milk obtained from the K-25/Grassy Creek area (CRM 14). The doses at CRM 14 and CRM 3.5 from milk ingestion alone are higher than the combined pathway of drinking water and milk ingestion because the time period over which a child could have been exposed through the combined pathways occurred only after the major releases of ^{131}I had ended. The highest concentration of ^{131}I in the Clinch River occurred in 1946, so this analysis was performed for a child born in that year. However, the drinking water intake for the Kingston City Municipal Water Supply was not moved to the Tennessee River (where it might have received contamination from the Clinch River) until 1955 (Section 7).

Table 13.6 Thyroid dose (cSv) from ingestion of ^{131}I by a female child from age 0 to 14.

		95% Subjective Confidence Interval		
Pathway and Location	**Time Period**	**lower limit**	**central value**	**upper limit**
Milk at CRM 14	1946 - 1960	0.00058	0.0062	0.054
Milk at CRM 3.5	1946 - 1960	0.00055	0.0044	0.042
Water at CRM 0	1955 – 1969	0.000039	0.00031	0.0021
Milk at CRM 3.5 and water at CRM 0	1955 - 1969	0.00014	0.00091	0.0047

13.3 Estimates of Excess Lifetime Risk of Cancer Incidence

The organ-specific dose estimates summarized in Section 13.2 were used as the basis for organ-specific estimates of excess lifetime risk of cancer incidence. The methods used for risk estimation are described in Sections 4 and 12. In general, the dose-response functions were based on cancer incidence data for the A-bomb survivors and on the background incidence rates

of cancer for East Tennessee. The dose-response data for exposure to ^{131}I were based on numerous additional sources, including epidemiological investigations of children exposed to x-rays and children exposed to ^{131}I in the aftermath of the Chernobyl accident.

Although radiation doses were estimated for every organ for which information could be obtained (Sections 10 and 11), dose-response functions could not be developed for the entire set. For several organs (adrenals, bone, small intestine, upper large intestine, skin, spleen, testes, thymus, and uterus), either the A-bomb survivor data or the background cancer incidence in East Tennessee was not available (Section 12); in addition, risks were not calculated for organs such as the prostate for which doses were not estimated (due to absence of sufficient dosimetric information). The total risks described in this section were calculated from a tissue-weighted effective dose, using dose-response and background incidence information for total solid tumors and leukemia (Section 12.2.6).

To facilitate comparison and interpretation of the results in Section 13.3, the estimates of excess lifetime risk of cancer incidence for each endpoint are compared to a reference level of 1×10^{-4} (1 chance in 10,000) lifetime risk. Use of a reference level is not an attempt to decide the significance of these results for any individual, but is intended simply to assist readers in comparison of these results across endpoints or with the results of other studies. A reference level of 1×10^{-4} risk is below the current limits of epidemiological detection for all types of cancer for the Task 4 exposure situation. It is clearly below the lifetime risk level of 5×10^{-3} (based on a whole-body lifetime dose of 0.07 Sv) recommended for dose reconstruction by the National Research Council (NRC, 1995). In addition, a level of 1×10^{-4} is also consistent with a level of negligible risk recommended by the International Atomic Energy Agency (IAEA, 1986) and the National Council for Radiation Protection and Measurements (NCRP, 1993), with the thyroid dose of 1 rad used to define the limits of the domain of the Hanford Environmental Dose Reconstruction (Shleien, 1992), with the value established by the ORHASP for prioritizing resources in the Oak Ridge Dose Reconstruction (Section 3 of this report; Thiessen et al., 1996), and with decisions made regarding the need for environmental remediation in the Oak Ridge region (Levine et al., 1994; Jacobs, 1995).

13.3.1 Ingestion of Fish

Estimated organ-specific risks of cancer incidence for Category I fish consumers (1-2.5 meals per week) near Jones Island (CRM 20.5) ranged from 3.5×10^{-8} to 9.3×10^{-4}, with central values from 9.4×10^{-7} to 3.7×10^{-5} for males and from 6.8×10^{-7} to 5.5×10^{-5} for females (shown for selected organs in Table 13.7; a complete list of organ-specific values for excess lifetime risk of cancer incidence is provided in Appendix 13B). For the Kingston area (CRM 0), risk values ranged from 5.2×10^{-9} to 7.1×10^{-5}, with central values from 9.5×10^{-8} to 4.3×10^{-6} for males and from 8.4×10^{-8} to 6.5×10^{-6} for females (Table 13.7, Figure 13.2; Appendix 13B). As with the doses, the risks from consumption of fish from the Jones Island area are about a factor 8-9 higher than the risks for fish from the Kingston area, reflecting the differences in radionuclide concentrations between the two areas. Risks for Categories II and III (Table 13.7) are lower than those for Category I in proportion to the lower intake rates assumed for these individuals.

Table 13.7 Selected organ-specific risks[a] of cancer incidence from all radionuclides for consumption of fish from the Clinch River near Jones Island or Kingston[b].

Organ	Category I	Category II	Category III
		Jones Island (CRM 20.5)	
Males			
Lower large intestine	2.5×10^{-6} - 2.8×10^{-4} (2.7×10^{-5})	7.0×10^{-7} - 1.4×10^{-4} (1.1×10^{-5})	1.7×10^{-7} - 3.4×10^{-5} (2.5×10^{-6})
Red bone marrow	2.5×10^{-6} - 3.4×10^{-4} (3.7×10^{-5})	8.6×10^{-7} - 1.3×10^{-4} (1.6×10^{-5})	1.8×10^{-7} - 3.7×10^{-5} (3.8×10^{-6})
Females			
Breast	4.3×10^{-6} - 9.3×10^{-4} (5.5×10^{-5})	1.9×10^{-6} - 4.1×10^{-4} (2.3×10^{-5})	3.7×10^{-7} - 1.0×10^{-4} (5.6×10^{-6})
Lower large intestine	1.6×10^{-6} - 2.6×10^{-4} (2.2×10^{-5})	5.8×10^{-7} - 1.4×10^{-4} (9.3×10^{-6})	1.4×10^{-7} - 3.2×10^{-5} (2.0×10^{-6})
Red bone marrow	2.5×10^{-6} - 4.0×10^{-4} (2.8×10^{-5})	8.6×10^{-7} - 1.8×10^{-4} (1.2×10^{-5})	1.6×10^{-7} -4.9×10^{-5} (2.7×10^{-6})
		Kingston (CRM 0)	
Males			
Lower large intestine	3.4×10^{-7} - 3.1×10^{-5} (3.2×10^{-6})	1.0×10^{-7} - 1.2×10^{-5} (1.4×10^{-6})	2.9×10^{-8} - 3.3×10^{-6} (3.2×10^{-7})
Red bone marrow	5.4×10^{-7} - 2.8×10^{-5} (4.3×10^{-6})	1.5×10^{-7} - 1.2×10^{-5} (1.9×10^{-6})	3.5×10^{-8} - 3.3×10^{-6} (4.4×10^{-7})
Females			
Breast	9.1×10^{-7} - 7.1×10^{-5} (6.5×10^{-6})	2.9×10^{-7} - 3.3×10^{-5} (2.8×10^{-6})	5.2×10^{-8} - 8.4×10^{-6} (6.6×10^{-7})
Lower large intestine	3.4×10^{-7} - 2.3×10^{-5} (2.8×10^{-6})	8.9×10^{-8} - 1.1×10^{-5} (1.2×10^{-6})	2.5×10^{-8} - 2.9×10^{-6} (2.6×10^{-7})
Red bone marrow	4.3×10^{-7} - 2.3×10^{-5} (3.5×10^{-6})	1.5×10^{-7} - 1.1×10^{-5} (1.5×10^{-6})	3.5×10^{-8} - 3.2×10^{-6} (3.3×10^{-7})

Footnotes to Table 13.7

[a] Ranges indicate the 95% subjective confidence intervals. Values in parentheses are the 50th percentiles.

[b] Consumption rates are as follows:

Category I, 1-2.5 meals per week, or 7.1 – 33 kg y^{-1} for males and 5.7 – 27 kg y^{-1} for females.

Category II, 0.25 – 1.3 meals per week, or 2.2 – 16 kg y^{-1} for males and 1.8 – 14 kg y^{-1} for females.

Category III, 0.04 – 0.33 meals per week, or 0.39 – 4.3 kg y^{-1} for males and 0.32 – 3.6 kg y^{-1} for females.

For all categories, it was assumed that 20 to 100% of the fish was contaminated, and 80 to 90% of the radioactivity in the fish was retained after processing (Sections 7, 13.1).

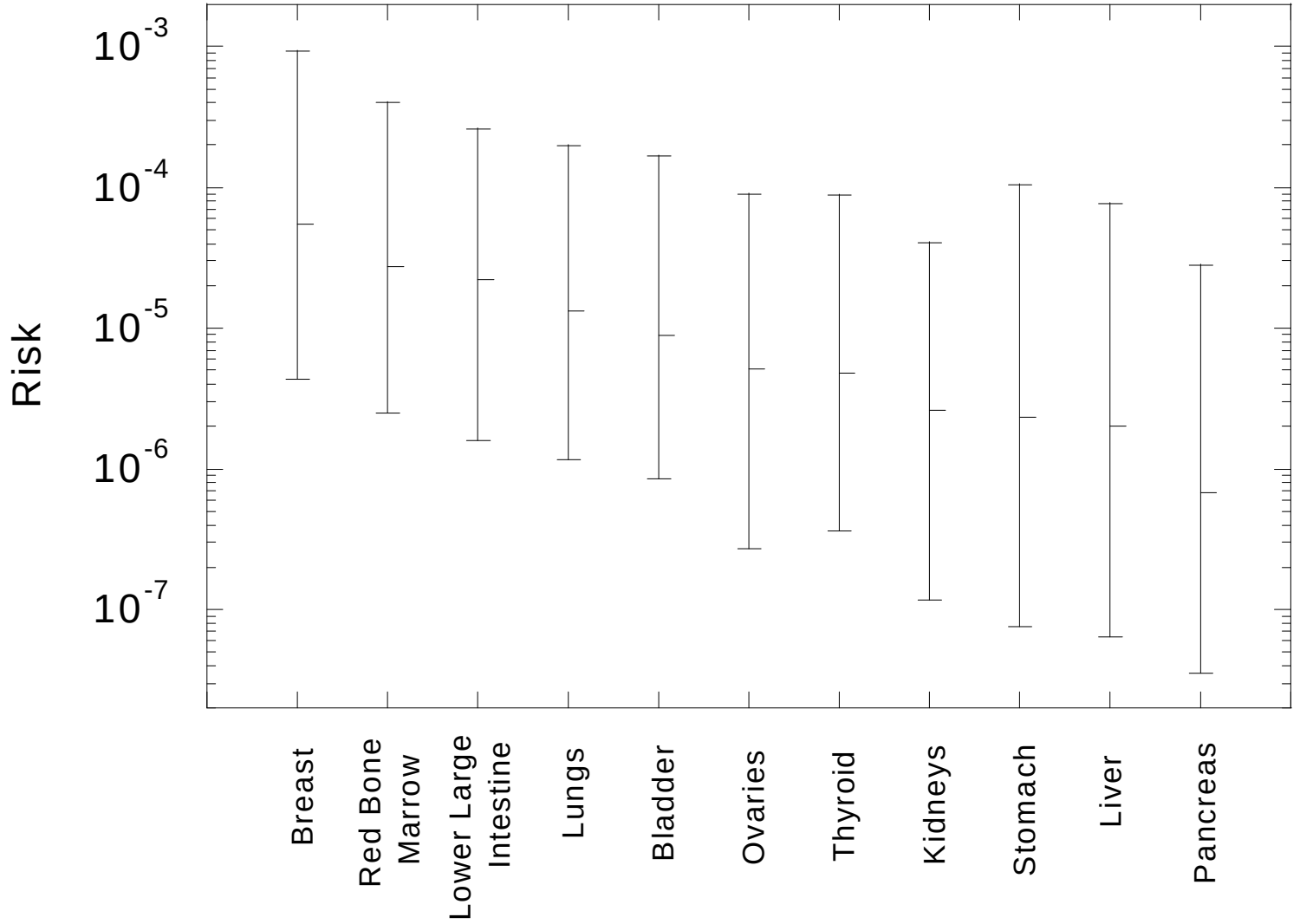

Figure 13.2 Organ-specific risks for females from ingestion of fish at CRM 20.5 (Category I consumers, 1-2.5 meals per week). The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50th percentiles) are also indicated.

Although the bone marrow received among the highest doses and the breast among the lowest doses, the breast has the higher risk by almost a factor of 2 (Table 13.7). For females, the highest risks are for breast and red bone marrow; for males, the highest risk is for the red bone marrow. The lowest risks for males or females were for the pancreas. The difference between the highest and lowest risks for any one location and consumption rate is about a factor of 70-80 for females and 40 for males, although the differences in doses are only a factor of 2-4 (Section 13.2.1). This situation illustrates the great difference in organ sensitivities to radiation-induced cancer and underlines the importance of calculating risks as well as doses.

For Category I consumers near Jones Island (CRM 20.5), the 95% subjective confidence intervals for the total excess lifetime risk of cancer incidence for all radionuclides and organs included in this analysis was 3.6×10^{-5} to 3.5×10^{-3} (50^{th} percentile, 2.8×10^{-4}) for males and 2.9×10^{-5} to 2.8×10^{-3} (50^{th} percentile, 2.3×10^{-4}) for females (Table 13.8; Appendix 13C). The slightly higher total risk for males is due primarily to the difference in ingestion rates (Section 7). For both males and females, the largest contribution to the total risk (about 90%) is from ^{137}Cs (Table 13.8), which is concentrated in fish to a much larger extent than are the other radionuclides (Section 8). The total risks to Category I consumers at Kingston (CRM 0) are about a factor of 7-9 lower than at Jones Island (Table 13.8; Appendix 13C), once again reflecting the differences in radionuclide concentrations in fish at the two locations. However, even at Kingston, the upper bound risk for Category 1 consumers is still above a reference level of 1×10^{-4} for both males and females (Table 13.8; Figure 13.3).

Values of total excess lifetime risk of cancer (central values) are about a factor of 2-3 higher for Category I consumers than for Category II consumers at any given location and about a factor of 10 higher for Category I consumers than for Category III consumers for any given location (Figure 13.3; Appendix 13C). This finding is expected, based on the relative differences in the amount of fish consumed by individuals in each category. At Jones Island (CRM 20.5), the risks (central values) for the Category II consumers reach or exceed a reference level of 1×10^{-4}, although the upper 95% subjective confidence limits for Category II consumers (male and female) approach or exceed this value at all locations (Figure 13.3; Appendix 13C). For Category III (male and female), the upper 95% subjective confidence limit on the total risk estimate is below a reference level of 1×10^{-4} for all locations except Jones Island (Figure 13.3; Appendix 13C).

For the Jones Island area (CRM 20.5), the large total risk from ingestion of fish for the Category I consumer is considered by the study team to be a conservative estimate, because the likelihood is small that someone consumed that much fish from only the Jones Island area. Based on the number of tenant farmers in the area during the 1940s and 1950s (the period of the greatest potential exposures), probably at most 100 individuals might have actually been exposed at that level (Section 7). The number of tenant farmers living in Loudon and Roane Counties was approximately 300 and 400, respectively, in 1940, and 100 and 200, respectively, in 1950 (USDOC, 1942). However, only a few of these tenant farmers would have lived in the Jones Island area.

Table 13.8 Estimates of total excess lifetime risk of cancer incidence by radionuclide for Category I consumers[a] of fish at CRM 20.5 and CRM 0 (fish ingestion only).

Radionuclide	**95% Subjective Confidence Interval**		
	lower limit	**central value**	**upper limit**
CRM 20.5, male			
^{137}Cs	3.0×10^{-5}	2.6×10^{-4}	3.4×10^{-3}
^{106}Ru	9.8×10^{-8}	1.7×10^{-6}	3.0×10^{-5}
^{90}Sr	6.8×10^{-7}	1.1×10^{-5}	1.2×10^{-4}
^{60}Co	6.6×10^{-8}	9.8×10^{-7}	1.4×10^{-5}
Total Risk[b]	3.6×10^{-5}	2.8×10^{-4}	3.5×10^{-3}
CRM 20.5, female			
^{137}Cs	2.4×10^{-5}	2.2×10^{-4}	2.8×10^{-3}
^{106}Ru	8.1×10^{-8}	1.4×10^{-6}	2.5×10^{-5}
^{90}Sr	5.5×10^{-7}	9.0×10^{-6}	1.0×10^{-4}
^{60}Co	5.5×10^{-8}	8.1×10^{-7}	1.1×10^{-5}
Total Risk[b]	2.9×10^{-5}	2.3×10^{-4}	2.8×10^{-3}
CRM 0, male			
^{137}Cs	4.4×10^{-6}	3.4×10^{-5}	2.6×10^{-4}
^{106}Ru	1.8×10^{-8}	2.8×10^{-7}	3.3×10^{-6}
^{90}Sr	1.1×10^{-7}	1.4×10^{-6}	1.3×10^{-5}
^{60}Co	1.4×10^{-8}	1.4×10^{-7}	1.4×10^{-6}
Total Risk[b]	5.6×10^{-6}	3.7×10^{-5}	2.6×10^{-4}
CRM 0, female			
^{137}Cs	3.6×10^{-6}	2.8×10^{-5}	2.1×10^{-4}
^{106}Ru	1.5×10^{-8}	2.3×10^{-7}	2.7×10^{-6}
^{90}Sr	9.4×10^{-8}	1.1×10^{-6}	1.1×10^{-5}
^{60}Co	1.1×10^{-8}	1.1×10^{-7}	1.2×10^{-6}
Total Risk[b]	4.6×10^{-6}	3.0×10^{-5}	2.1×10^{-4}

[a] 1-2.5 meals per week (Sections 7, 13.1)

[b] The calculations were performed using Monte Carlo techniques to add the distributions. Therefore, the total risks may not equal the arithmetic sums of the components.

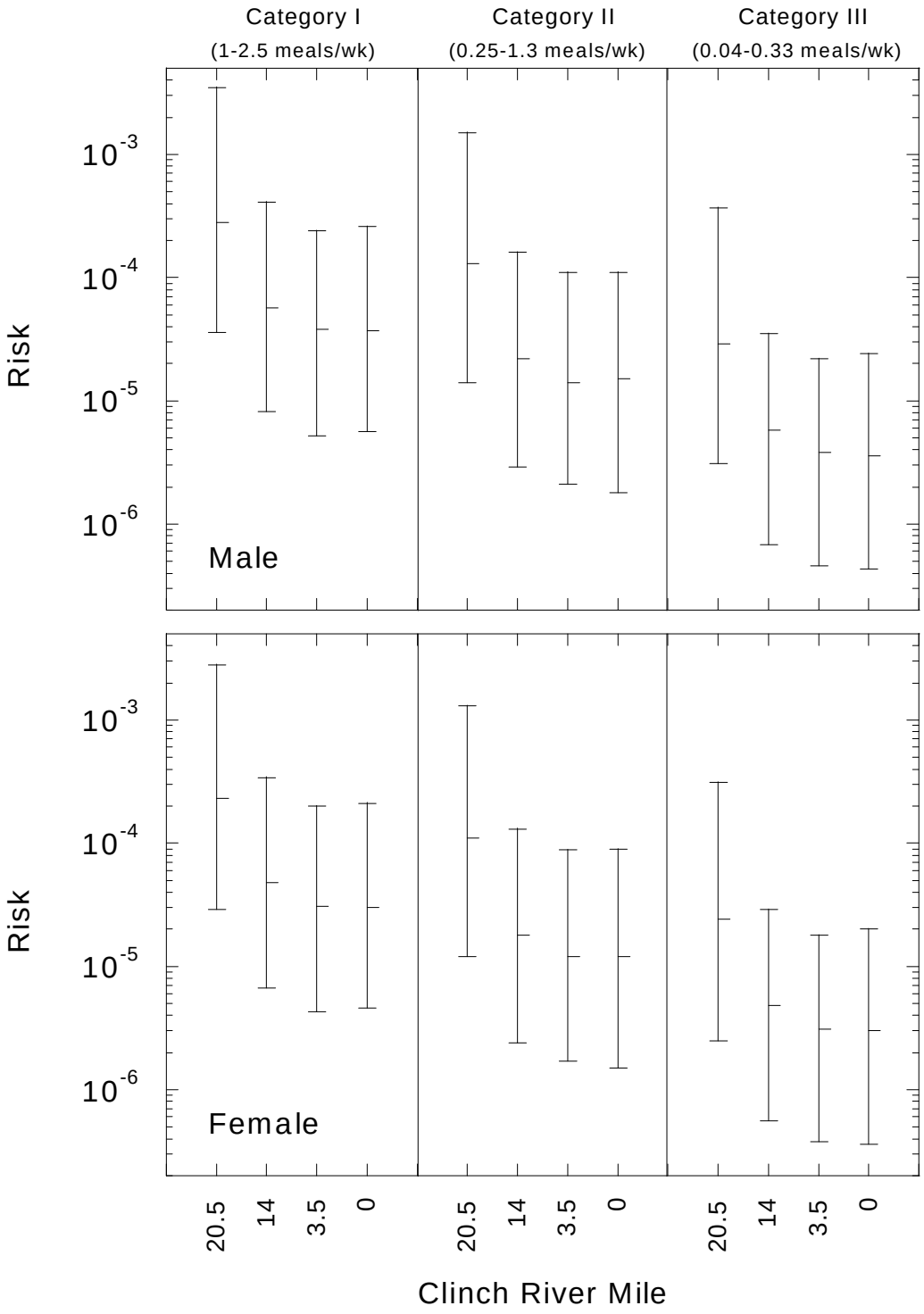

Figure 13.3 Excess lifetime risk of cancer incidence for male (top) and female (bottom) reference individuals consuming fish at each location of interest along the Clinch River. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50th percentiles) are indicated by dashes.

13.3.2 Other Exposure Pathways

Estimates of organ-specific and total excess lifetime risks of cancer incidence are summarized by pathway and radionuclide in Figures 13.4 –13.6 and Appendices 13-B and 13-C. For the fish ingestion and drinking water pathways, the excess lifetime risks of cancer incidence decrease as the downstream radionuclide concentration in water decreases. For ingestion of meat or milk and for the external exposure pathway, the risk estimates for the Jones Island area (CRM 20.5) are lower than at the downstream locations CRM 14 and CRM 3.5 (Appendix 13C). This discrepancy is explained by the time periods used in the calculations. Until the Melton Hill Dam went into operation in 1963, the plume from White Oak Creek was not mixed until well below CRM 20.5, and the public (south) bank of the river was considered to be uncontaminated. Therefore, the exposure scenario for these pathways at the Jones Island area (CRM 20.5) extends only from 1964 to 1991, after Melton Hill Dam became operational.

The highest potential for exposure to shoreline sediments occurred at the reference locations downstream from White Oak Creek (CRM 14, 3.5, and 0; Figure 13.4; Appendix 13C). The risks for all three areas are virtually the same, with a 95% subjective confidence interval of about 5×10^{-6} to 1×10^{-4}. The dominant contributors to the risks from external exposure are ^{60}Co and ^{137}Cs. The total lifetime risks from external exposure to shoreline sediments equal or exceed those from ingestion of fish by Category II and III consumers at all locations except CRM 20.5.

The 95% subjective confidence interval for the total excess lifetime risk of cancer incidence for the drinking water pathway at the K-25/Grassy Creek area (CRM 14) is 3.5×10^{-6} to 4.6×10^{-5} (50^{th} percentile, 1.2×10^{-5}; Table 13.9; Figure 13.4). Slightly lower risks were obtained for the Kingston Steam Plant area (CRM 3.5; Figure 13.4), but substantially lower risks were estimated for those exposed only from the Kingston City Municipal Water Supply (CRM 0; Figure 13.4; Appendix 13C). Although the estimates for Kingston (CRM 0) reflect a higher usage, based on a residential as opposed to occupational scenario, they also reflect a lower level of water contamination than for CRM 14 or 3.5. The drinking water intake for the city of Kingston is actually on the Tennessee River, and contamination occurs only through backflow from the Clinch River into the Tennessee River (Section 7). Additionally, exposure at Kingston could have occurred only after 1955 (Section 7). Total risks from drinking water at CRM 14 or 3.5 are comparable to the upper half of the 95% subjective confidence interval for the total risk estimate for Category III consumers of fish. The dominant contributors to risk at all locations are ^{106}Ru and ^{90}Sr.

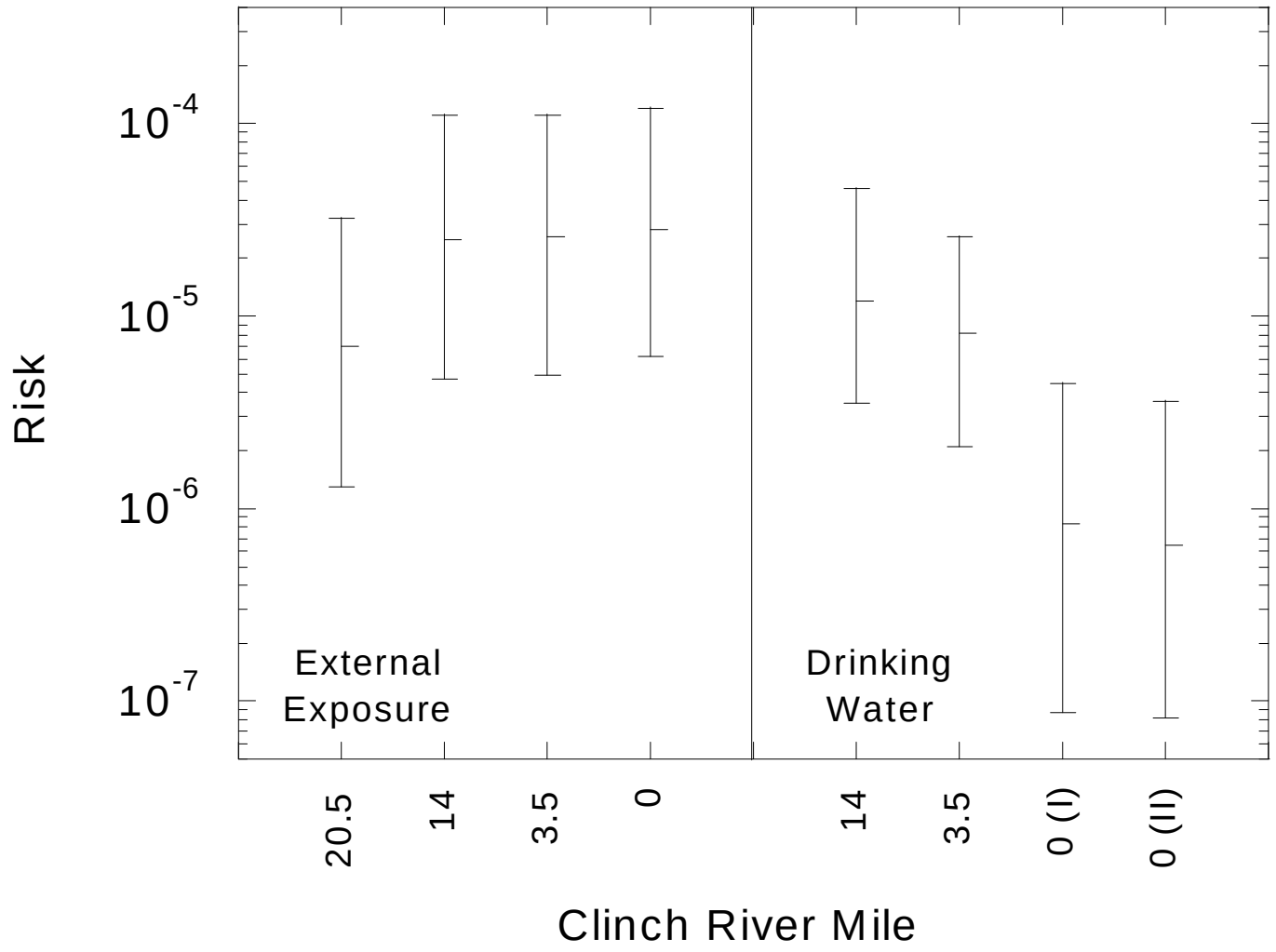

Figure 13.4 Excess lifetime risk of cancer by location for the external exposure and drinking water pathways. The vertical lines indicate the 95% subjective confidence intervals on the estimate risks; the central values (50^{th} percentiles) are also indicated. For drinking water at CRM 0, the Categories I and II refer to residential and transient use, respectively (Section 7).

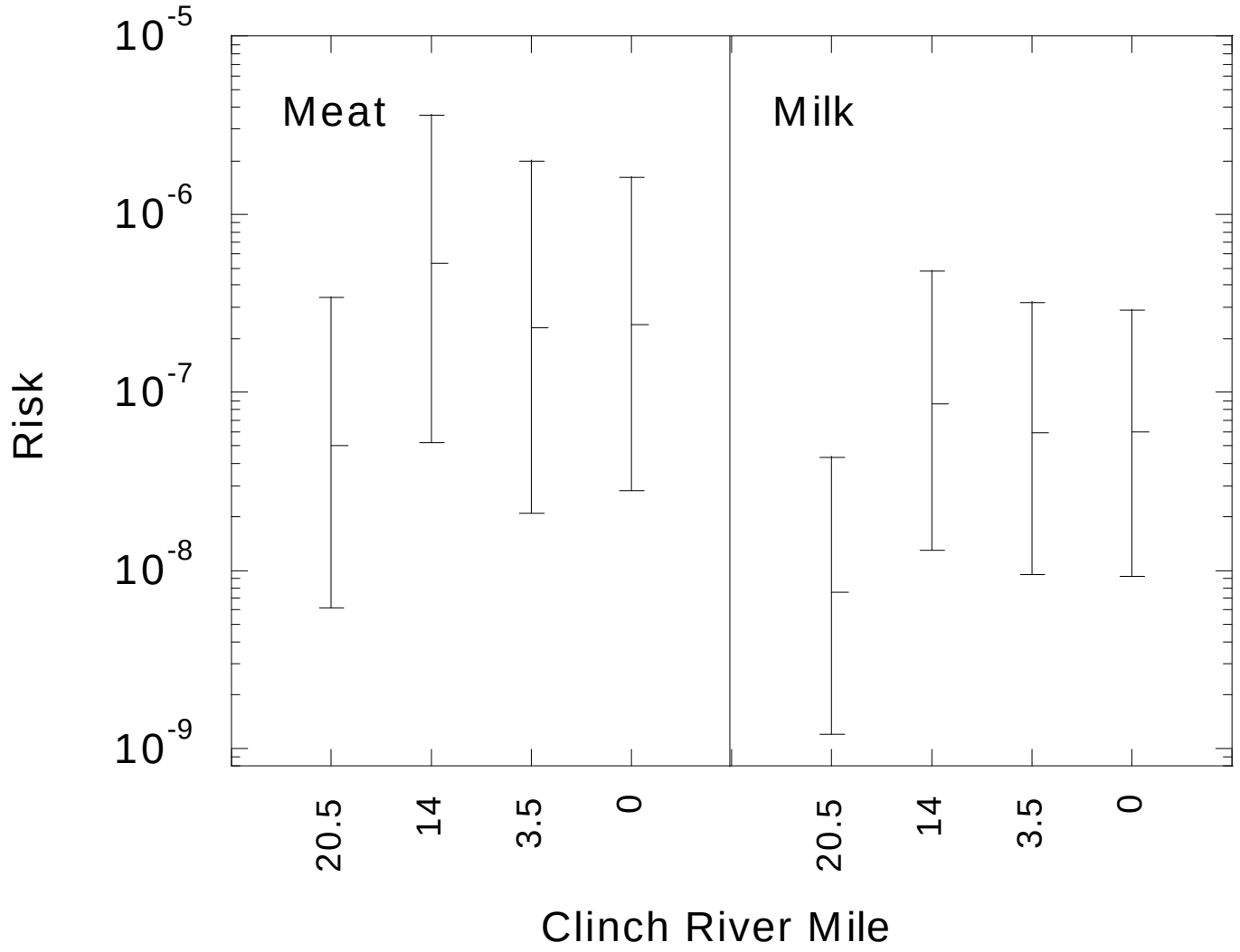

Figure 13.5 Excess lifetime risk of cancer by location for meat and milk ingestion pathways. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50^{th} percentiles) are also indicated.

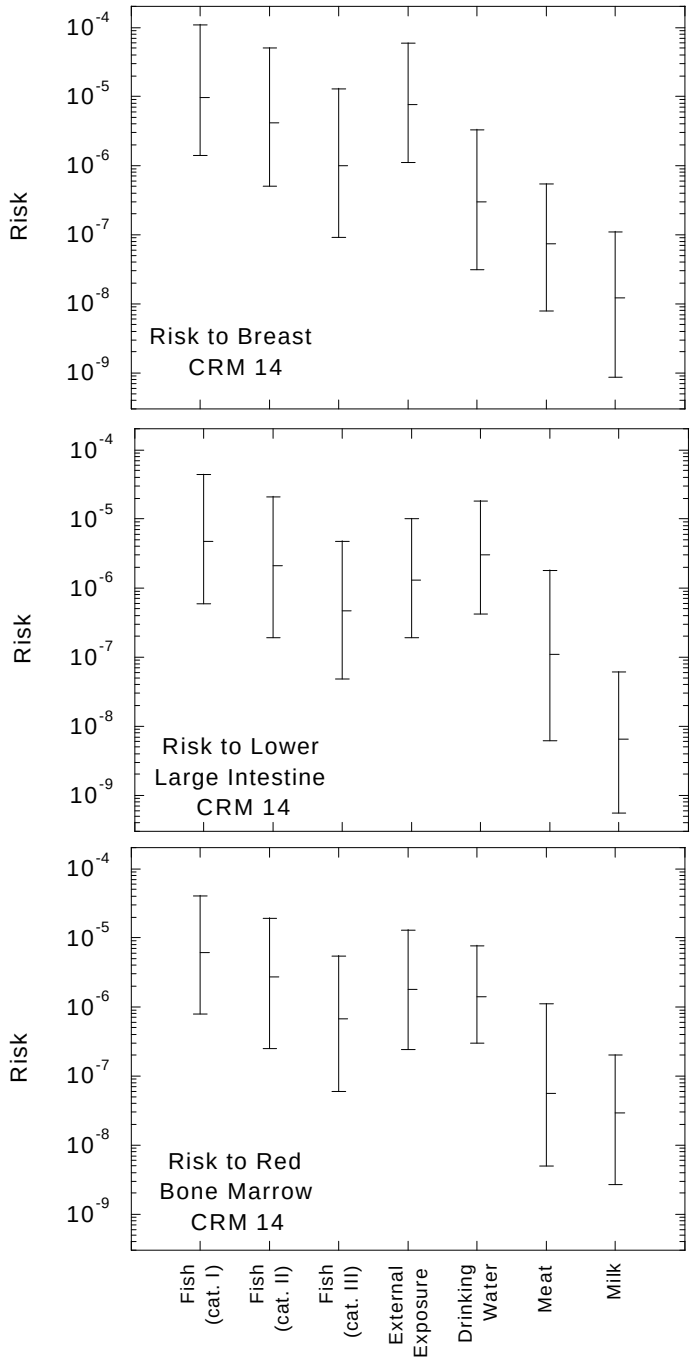

Figure 13.6 Organ-specific risks of cancer incidence for three primary organs of interest (breast for females, lower large intestine for males, and red bone marrow for males) for each exposure pathway at CRM 14. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50th percentiles) are also indicated. For fish consumption, the categories are as follows: Category I, 1-2.5 meals per week; Category II, 0.25 - 1.3 meals per week; Category III, 0.04 B 0.33 meals per week (Section 13.1).

Estimated organ-specific and total risks of cancer incidence from ingestion of meat and milk are significantly lower (1 or 2 orders of magnitude) than those estimated for fish ingestion by a Category III consumers and are even lower in comparison to the total risks for Categories I and II (Table 13.9; Figures 13.2-13.6; Appendix 13C). The lower bounds for the estimates of total risk from the ingestion of milk or meat are always below 10^{-7} risk (Figure 13.5), although the upper bounds for the estimates of total risk from the ingestion of meat exceed 10^{-6} for all locations except CRM 20.5. The ingestion of milk and meat for an adult are negligible contributors to the total excess lifetime risk of cancer incidence at every location.

Figure 13.6 provides a comparison by pathway for excess risk of cancer incidence for breast (females), lower large intestine (males), and red bone marrow (males) at CRM 14. For the breast, the risk from external exposure is higher than the risk for a Category III consumer of fish but is comparable for the Category I and II consumers. For the lower large intestine and red bone marrow, contributions from both external exposure and drinking water are important.

13.3.3 Combined Exposure Pathways

Total estimated risks of cancer incidence (all organs and radionuclides) were compared by pathway for each location of concern (Figures 13.7-13.10; Appendix 13C). The total risk for the combined fish ingestion, drinking water, and external exposure pathways were also calculated for each location (Figures 13.7-13.10; Appendix 13D). The "worst-case" combination (fish ingestion at CRM 20.5, drinking water and external exposure at CRM 14) was also examined (Figure 13.11; Appendix 13D). Risks from meat and milk ingestion are included in Figures 13.7-13.11, but these did not contribute to the total risk across pathways. The combined risks by location and fish ingestion rate are summarized in Figure 13.12 for males (females would have slightly lower risks due to the lower ingestion rate for fish).

At all locations, fish ingestion is the dominant contributor to risk for Category I consumers. However, except for the Jones Island area (CRM 20.5), external exposure is at least as important as fish ingestion for Category II and III consumers; drinking water contributes as well for CRM 14 and CRM 3.5. For all locations and ingestion rates, the upper bound of the estimated total risk exceeds a reference level of 1×10^{-4} (Figure 13.12). However, only for consumption of fish from CRM 20.5 does the central value of the total risk estimate exceed a reference level of 1×10^{-4} for the Category I and II consumers.

Table 13.9 Estimates of total excess lifetime risk of cancer incidence by exposure pathway and by radionuclide for the K-25/Grassy Creek area (CRM 14).

Radionuclide	95% Subjective Confidence Interval		
	lower limit	central value	upper limit
Ingestion of fish (Category I, male)[a]			
^{137}Cs	6.7×10^{-6}	5.1×10^{-5}	4.0×10^{-4}
^{106}Ru	2.6×10^{-8}	3.4×10^{-7}	3.5×10^{-6}
^{90}Sr	1.6×10^{-7}	1.9×10^{-6}	1.9×10^{-5}
^{60}Co	2.0×10^{-8}	1.9×10^{-7}	1.6×10^{-6}
All radionuclides[b]	8.2×10^{-6}	5.7×10^{-5}	4.1×10^{-4}
Ingestion of fish (Category III, female)[c]			
^{137}Cs	5.2×10^{-7}	4.4×10^{-6}	2.8×10^{-5}
^{106}Ru	1.7×10^{-9}	2.6×10^{-8}	2.9×10^{-7}
^{90}Sr	9.9×10^{-9}	1.3×10^{-7}	1.7×10^{-6}
^{60}Co	1.3×10^{-9}	1.4×10^{-8}	1.4×10^{-7}
All radionuclides[b]	5.6×10^{-7}	4.8×10^{-6}	2.9×10^{-5}
External exposure			
^{137}Cs	3.2×10^{-6}	1.7×10^{-5}	7.6×10^{-5}
^{106}Ru	5.3×10^{-8}	2.6×10^{-7}	1.2×10^{-6}
^{90}Sr	2.8×10^{-10}	1.4×10^{-9}	6.3×10^{-9}
^{60}Co	1.4×10^{-6}	7.4×10^{-6}	3.4×10^{-5}
^{144}Ce	8.6×10^{-9}	4.3×10^{-8}	2.0×10^{-7}
^{95}Zr	3.7×10^{-8}	1.8×10^{-7}	8.7×10^{-7}
^{95}Nb	3.1×10^{-9}	1.6×10^{-8}	9.9×10^{-8}
All radionuclides[b]	4.7×10^{-6}	2.5×10^{-5}	1.1×10^{-4}
Ingestion of drinking water			
^{137}Cs	5.1×10^{-8}	2.2×10^{-7}	9.7×10^{-7}
^{106}Ru	1.7×10^{-6}	7.1×10^{-6}	3.3×10^{-5}
^{90}Sr	9.0×10^{-7}	3.9×10^{-6}	1.8×10^{-5}
All radionuclides[b]	3.5×10^{-6}	1.2×10^{-5}	4.6×10^{-5}
Ingestion of meat			
^{137}Cs	2.7×10^{-8}	1.8×10^{-7}	1.2×10^{-6}
^{106}Ru	3.7×10^{-9}	1.7×10^{-7}	2.2×10^{-6}
^{90}Sr	7.8×10^{-10}	3.0×10^{-8}	8.2×10^{-7}
^{60}Co	7.0×10^{-12}	2.4×10^{-10}	7.0×10^{-9}
All radionuclides[b]	5.2×10^{-8}	5.3×10^{-7}	3.6×10^{-6}
Ingestion of milk			
^{137}Cs	3.1×10^{-9}	4.2×10^{-8}	3.4×10^{-7}
^{106}Ru	4.5×10^{-9}	2.9×10^{-8}	2.4×10^{-7}
^{90}Sr	1.5×10^{-11}	4.0×10^{-10}	6.3×10^{-9}
All radionuclides[b]	1.3×10^{-8}	8.6×10^{-8}	4.8×10^{-7}

[a] 1-2.5 meals per week, or 7.1-33 kg y^{-1}.

[b] The calculations were performed using Monte Carlo techniques to add the distributions. Therefore, the total risks may not equal the arithmetic sums of the components.

[c] 0.04-0.33 meals per week, or 0.32-3.6 kg y^{-1}.

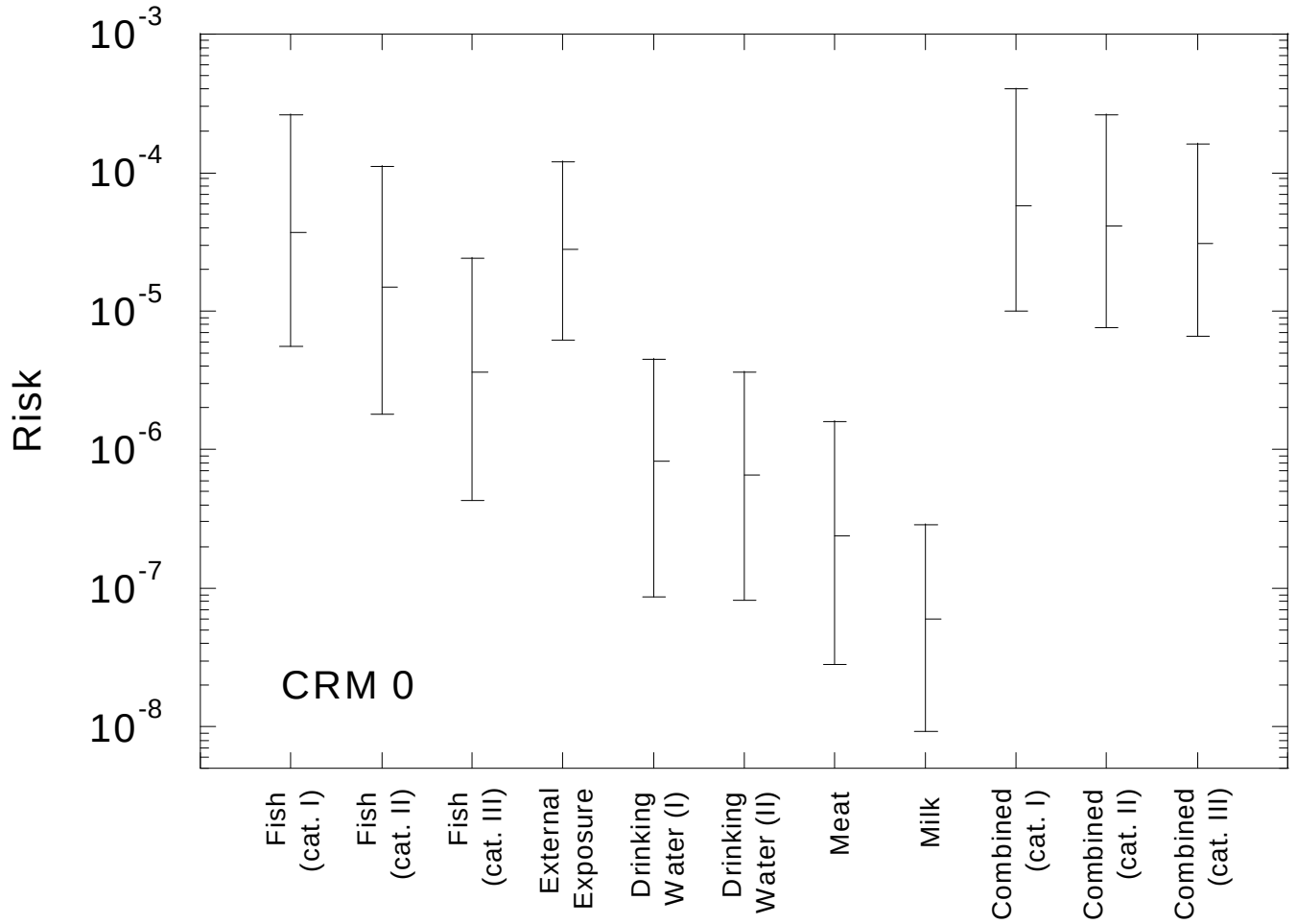

Figure 13.7 Estimates of total excess lifetime risk of cancer incidence for each pathway and for combinations of pathways for adult males at CRM 0. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50^{th} percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week. Drinking water I refers to residential water use in Kingston and drinking water II to transient water use in Kingston (Section 7). The combined risks refer to the sum of the risks from fish ingestion (Categories I, II, or III, as indicated), external exposure, and drinking water (Drinking water I, or residential use, in all cases).

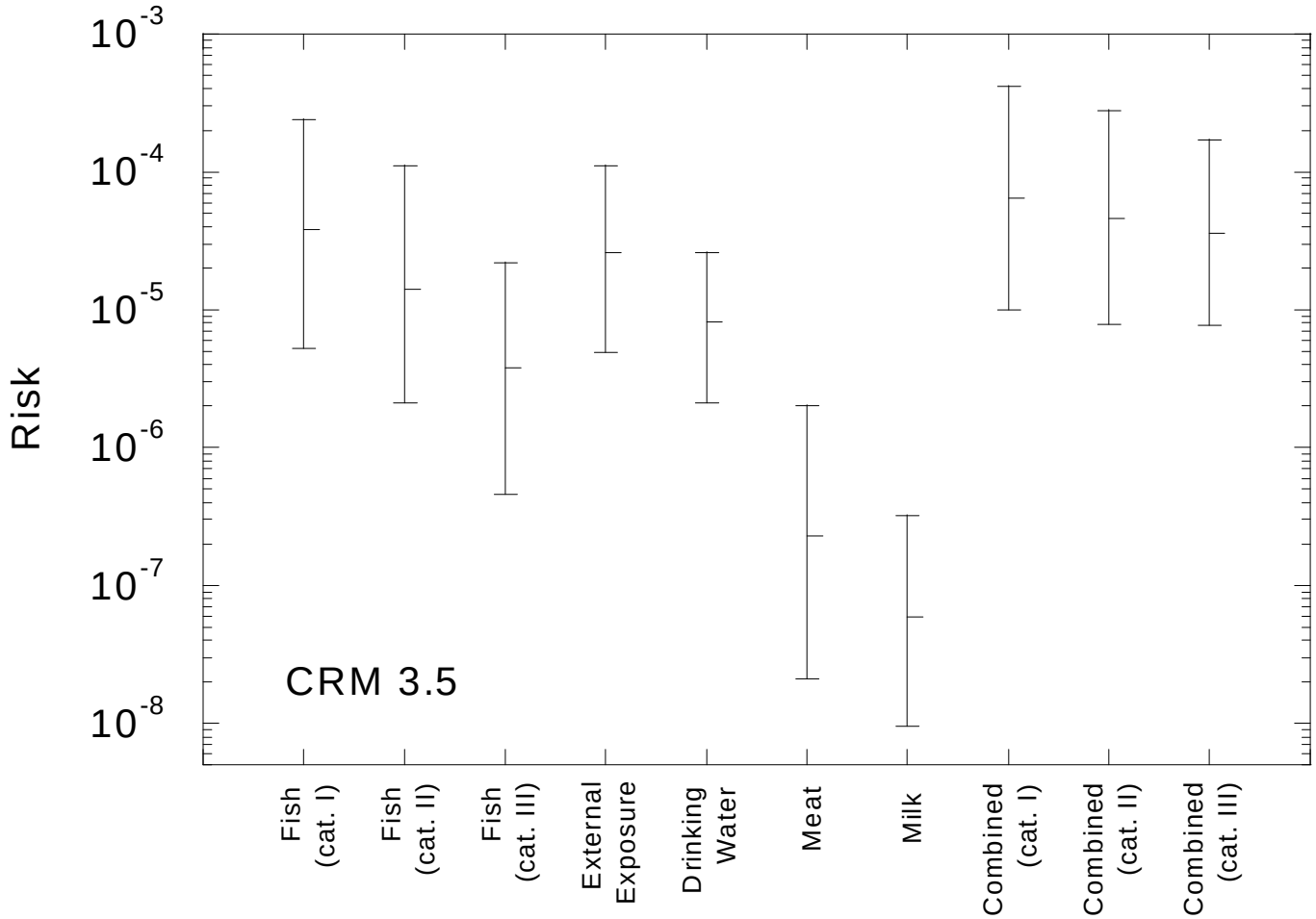

Figure 13.8 Estimates of total excess lifetime risk of cancer incidence for each pathway and for combinations of pathways for adult males at CRM 3.5. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50^{th} percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week. The combined risks refer to the sum of the risks from fish ingestion (Categories I, II, or III, as indicated), external exposure, and drinking water.

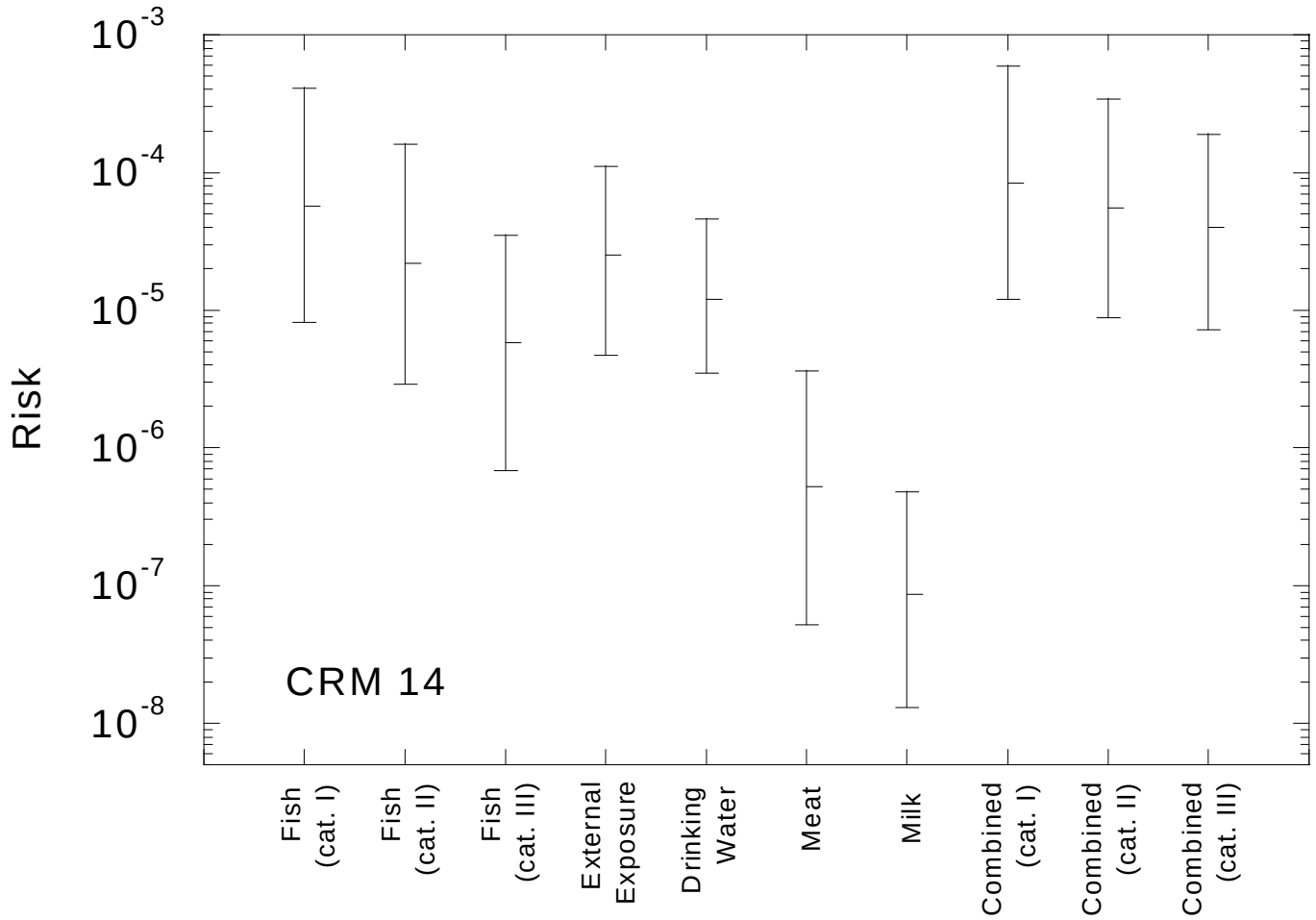

Figure 13.9 Estimates of total excess lifetime risk of cancer incidence for each pathway and for combinations of pathways for adult males at CRM 14. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50th percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week. The combined risks refer to the sum of the risks from fish ingestion (Categories I, II, or III, as indicated), external exposure, and drinking water.

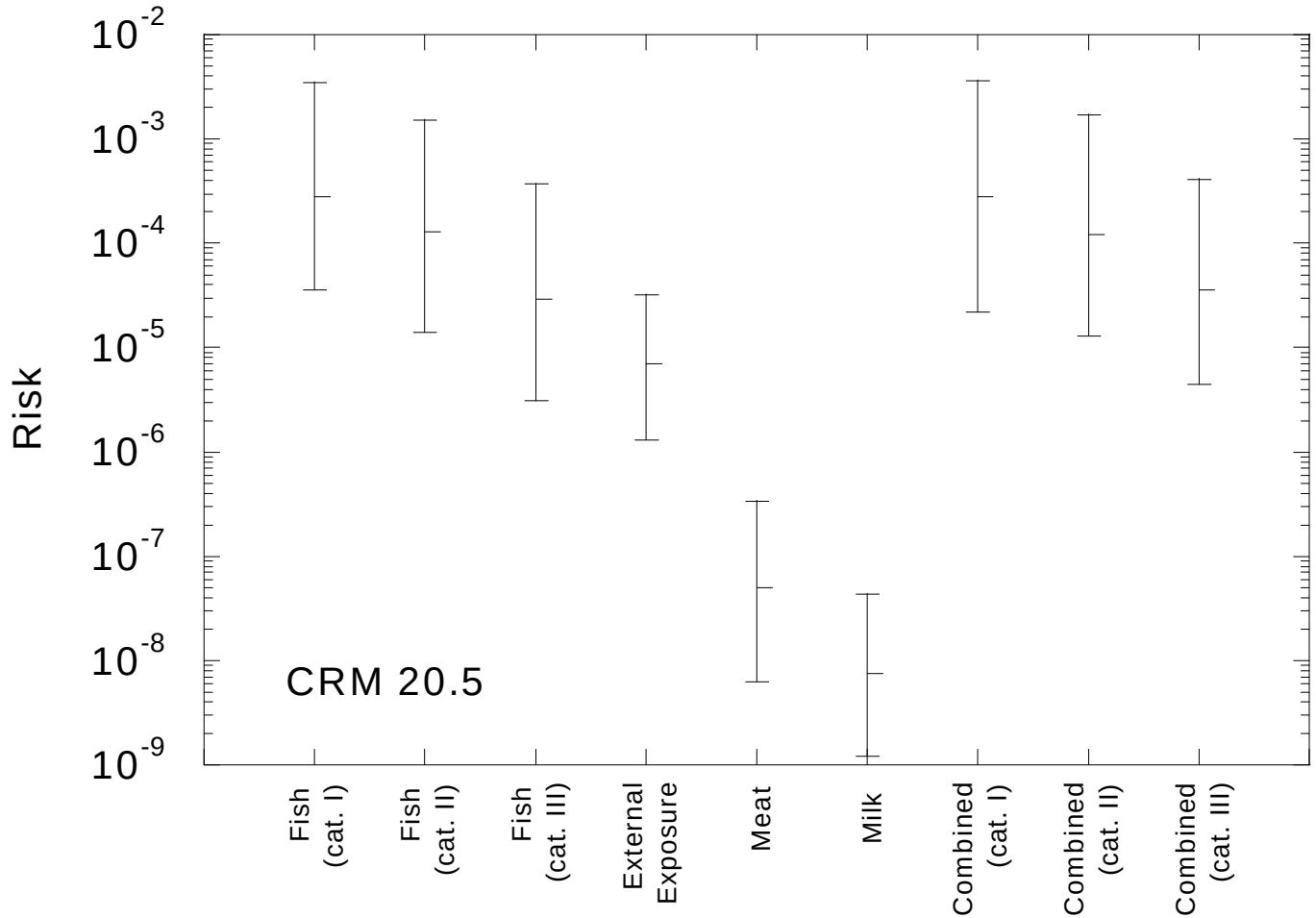

Figure 13.10 Estimates of total excess lifetime risk of cancer incidence for each pathway and for combinations of pathways for adult males at CRM 20.5. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50^{th} percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week. The combined risks refer to the sum of the risks from fish ingestion (Categories I, II, or III, as indicated) and external exposure.

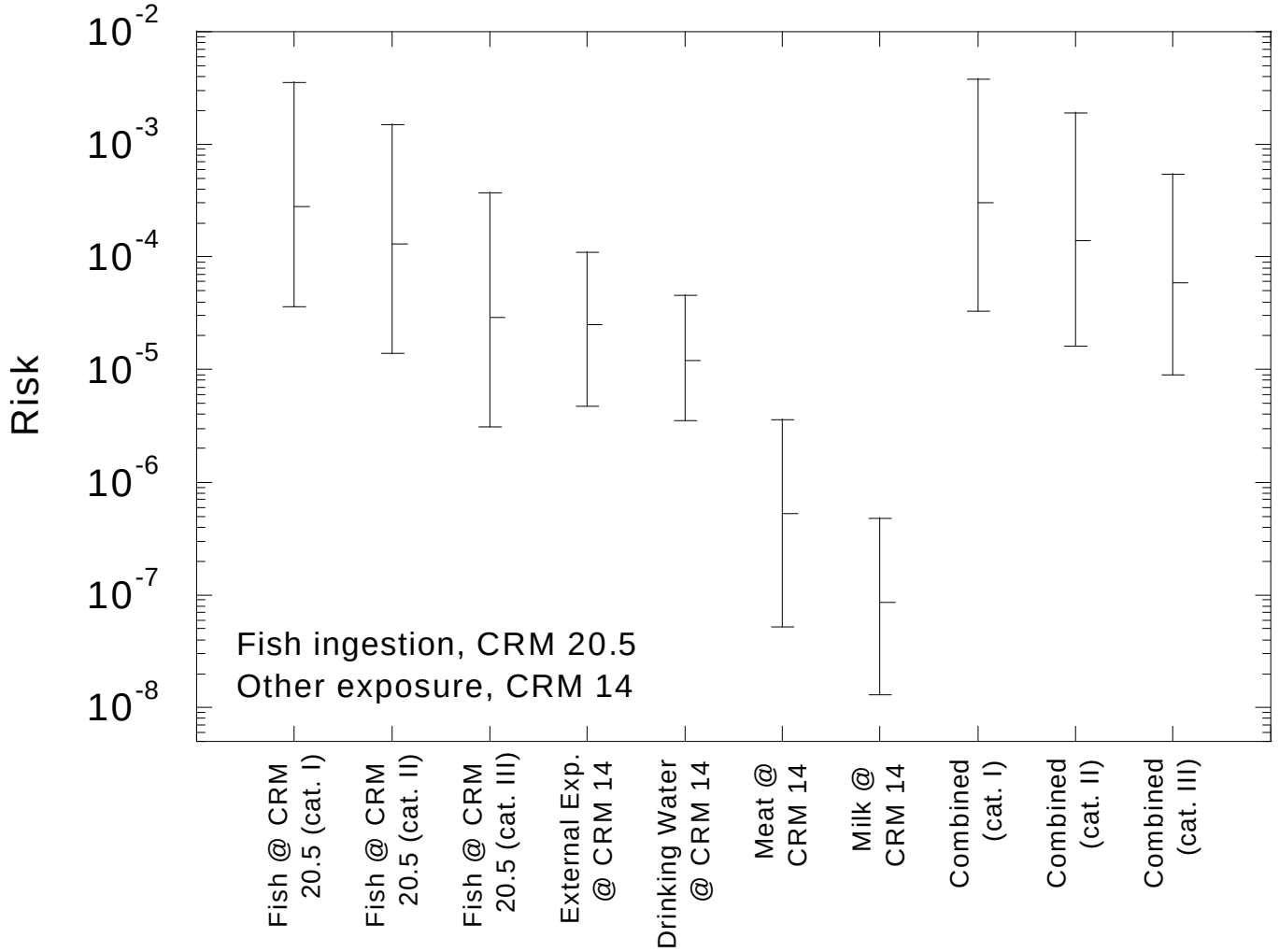

Figure 13.11 Estimates of total excess lifetime risk of cancer incidence for each pathway and for combinations of pathways for adult males, shown for fish ingestion at CRM 20.5 and for other pathways at CRM 14. The vertical lines indicate the 95% subjective confidence intervals on the estimate risks; the central values (50th percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week. The combined risks refer to the sum of risks from fish ingestion at CRM 20.5 (Categories I, II, or III as indicated) and external exposure and drinking water at CRM 14.

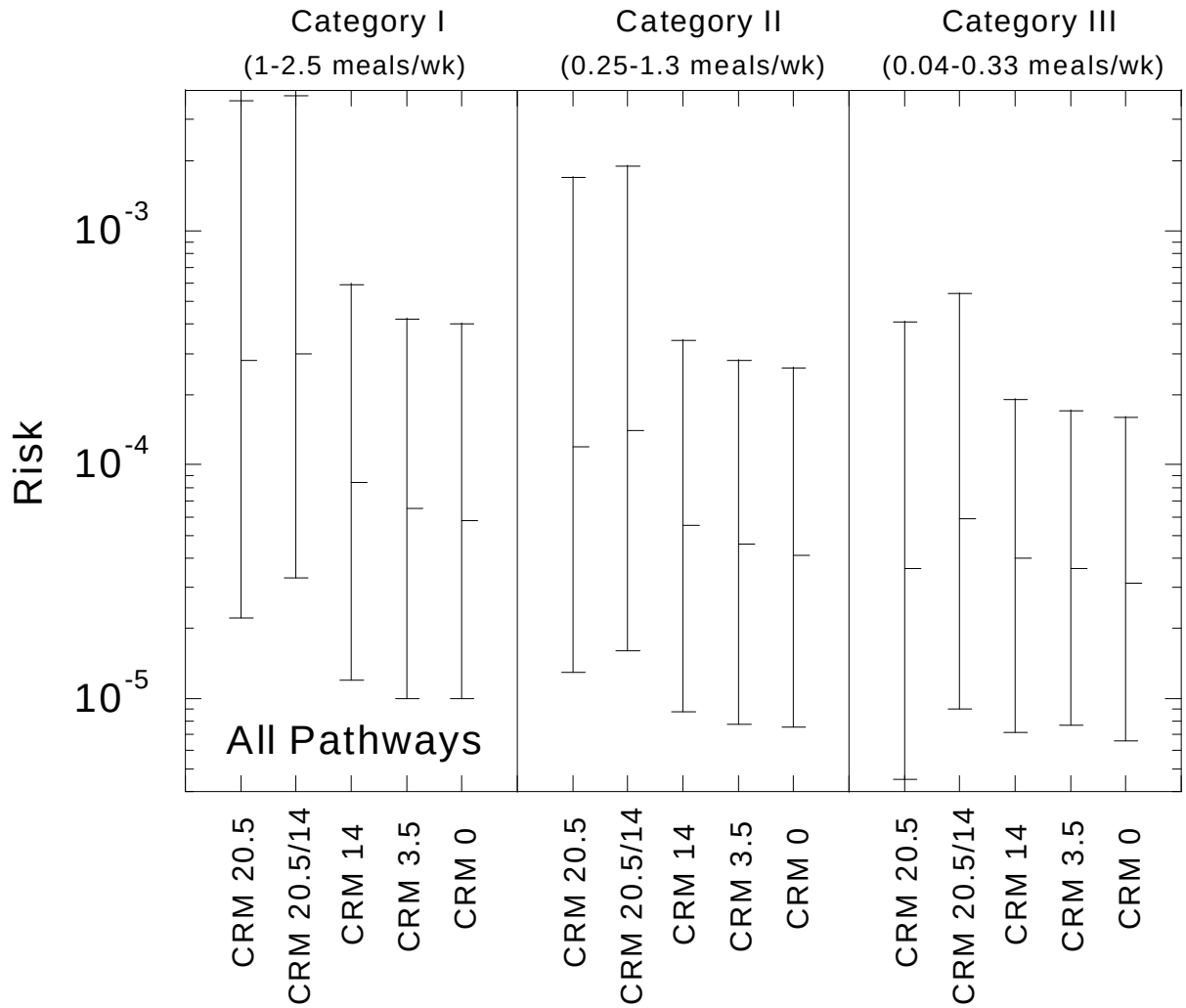

Figure 13.12 Estimates of total excess lifetime risk for adult males, shown by location for fish ingestion (Categories I, II, and III, as indicated), external exposure, and drinking water pathways combined. "CRM 20.5/14" indicates consumption of fish from CRM 20.5 and other exposure at CRM 14. Drinking water exposure at CRM 0 is based on the residential scenario (Section 7). The lines indicate the 95% subjective confidence intervals on the estimated risks; the central values (50^{th} percentiles) are also indicated.

13.3.4 Estimates of excess lifetime risk of thyroid cancer for a female child exposed from the drinking water and milk ingestion pathways

Estimated doses to the thyroid of a female child were described in Section 13.2.5. Although the thyroid doses reach about 0.054 cSv (upper 95% confidence limit for a female child in the K-25/Grassy Creek area, 1946-1960), the upper 95% confidence limits of the estimated excess lifetime risks of thyroid cancer incidence to a female child do not exceed a reference level of 10^{-4} risk (Table 13.10).

Table 13.10 Excess lifetime risk of thyroid cancer incidence from ingestion of ^{131}I by a female child from age 0 to 14.

		95% Subjective Confidence Interval		
Pathway and Location	**Time Period**	**lower limit**	**central value**	**upper limit**
Milk at CRM 14	1946 - 1960	1.1×10^{-7}	1.8×10^{-6}	2.5×10^{-5}
Milk at CRM 3.5	1946 - 1960	7.4×10^{-8}	1.2×10^{-6}	1.7×10^{-5}
Water at CRM 0	1955 – 1969	5.7×10^{-9}	7.3×10^{-8}	7.1×10^{-7}
Milk at CRM 3.5 and Water at CRM 0	1955 – 1969	2.4×10^{-8}	2.4×10^{-7}	1.8×10^{-6}

The highest risk was estimated for a female child ingesting milk obtained from an area near K-25/Grassy Creek (CRM 14). The risks from ingestion of milk alone at CRM 14 and CRM 3.5 are higher than the combined pathway of drinking water and milk ingestion (Table 13.10) due to the time period considered for the exposure (Section 13.2.5). Higher concentrations of ^{131}I were evident in the first years of release from White Oak Dam.

The excess lifetime risks of thyroid cancer are based on the most recent epidemiologic data for cancer incidence induced by external exposure to x-rays and gamma-rays. In this analysis, the estimates of risk were obtained by assuming that ^{131}I is less than or equal to external radiation from x-rays and gamma-rays in its effectiveness in cancer induction. The effectiveness of ^{131}I was treated as an uncertain parameter with a subjective probability distribution resulting in a mean effectiveness factor ranging from 1.0 to 0.2 (Section 10). That is, the central estimate of risk for a given thyroid dose of ^{131}I is less than the risk for the same dose resulting from external exposure to x rays or gamma rays. However, preliminary results from an epidemiological study of a large population of children exposed to ^{131}I during the Chernobyl accident in 1986 indicate that there may be no difference in the effectiveness of inducing thyroid cancer between ingestion or inhalation of ^{131}I and external exposure to x-rays or gamma-rays. If ^{131}I were assumed to be as effective as x-rays and gamma-rays in inducing thyroid cancer, the mean value (not necessarily identical to the central value, or median, reported here) of the excess lifetime risk

of thyroid cancer would be about 1.6 times larger; the lower 95% subjective confidence limit for a female child at CRM 14 would be increased by a factor of about 1.8 to 2, but the upper bound in Table 13.10 would be increased by only a factor of 1.2 to 1.3.

13.4 Comparison of the Results by Decade

Because the types of operations at the X-10 site and hence the type and amounts of the consequent releases changed over time, the analysis of exposure was also performed by decades to examine the risks to reference individuals who were exposed for a 10-year period rather than for the entire 48-year period. To evaluate the differences in estimates of excess lifetime risk of cancer incidence for different time periods of exposure, estimates were made by decade for male consumers of fish (all three categories), for the drinking water pathway, and for external exposure to shoreline sediment for all four locations (Figures 13.13 to 13.16).

The first two decades of exposures (1944-1953 and 1954-1963) produced the highest risks (Figures 13.13 to 13.16). For the first decade, the ingestion of fish produces the largest risks; however, external exposure to shoreline sediments becomes increasingly important in the last three decades of exposure. For an individual exposed from 1974 to 1991, the largest risks are from external exposure to shoreline sediments. The increasing importance over time of external exposure to shoreline sediments most likely reflects the build-up of radionuclides in sediment as well as the general decrease in radionuclide concentrations in water and fish. Because the ingestion of fish and external exposure to shoreline sediments contribute most to the total excess lifetime risks of cancer incidence, ^{137}Cs is the dominant radionuclide in all decades.

Both drinking water and external exposure would have a greater relative importance for people with low fish consumption rates (Section 13.3.2). The risk from fish consumption for a Category III consumer is about a factor of 4-5 lower for each time period than for a Category II consumer (Figures 13.13 to 13.16), but the risks from drinking water and external exposure would not change. It should be noted that the most important contributor to the drinking water pathway was ^{106}Ru, which was released into the Clinch River in high quantities from about 1959-1965. For this time period, the total risk from drinking water would be slightly higher than the value shown for 1954-1963 (Figures 13.13 to 13.16) and (for CRM 3.5 and CRM 14) only a little lower than the risks from external exposure or from fish ingestion for a Category II consumer.

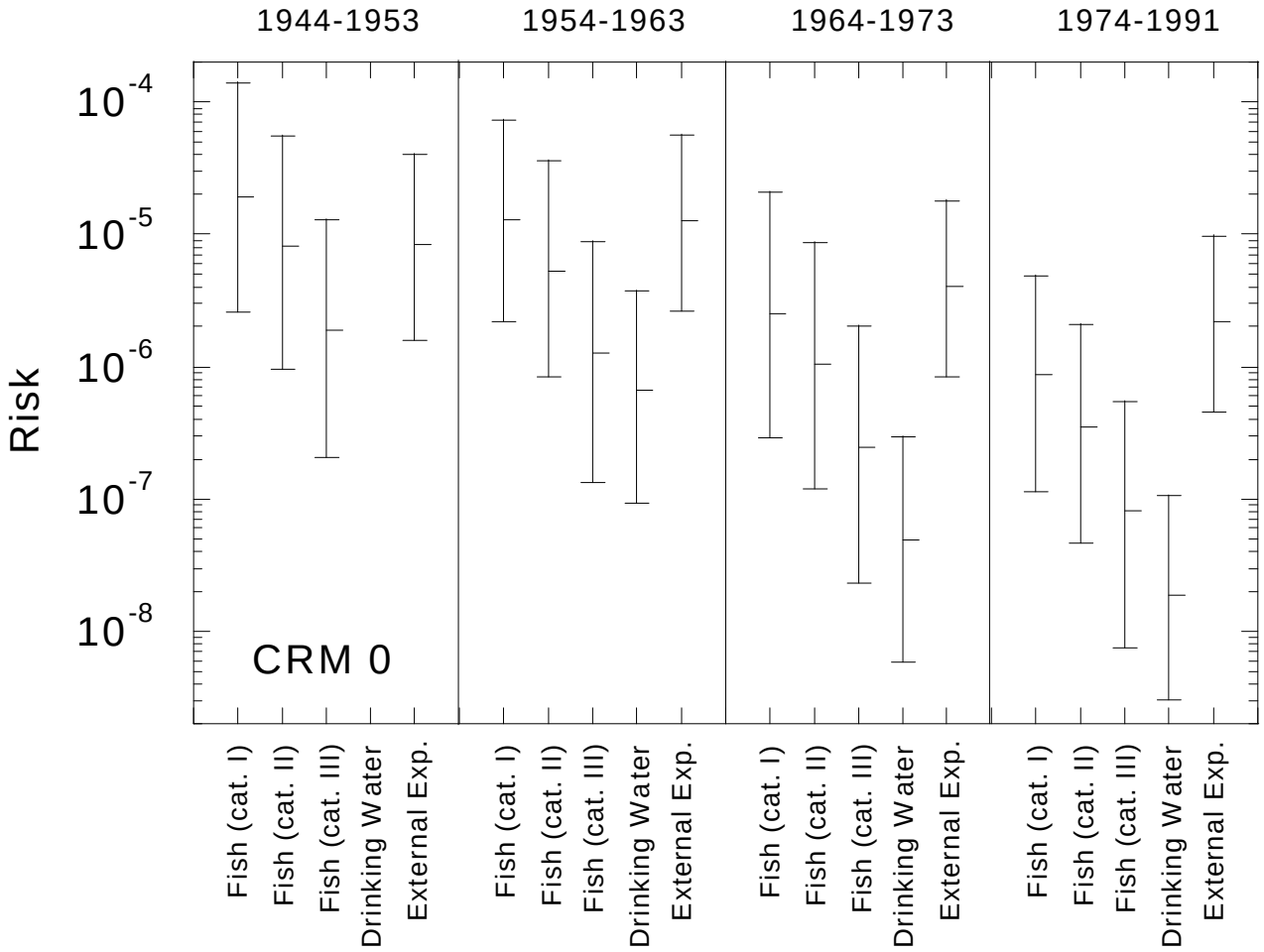

Figure 13.13 Estimates of total excess lifetime risk of cancer incidence for adult males, shown by exposure pathway at CRM 0 for four decades of interest. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; central values (50^{th} percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week. "Drinking water" refers to residential use in the city of Kingston.

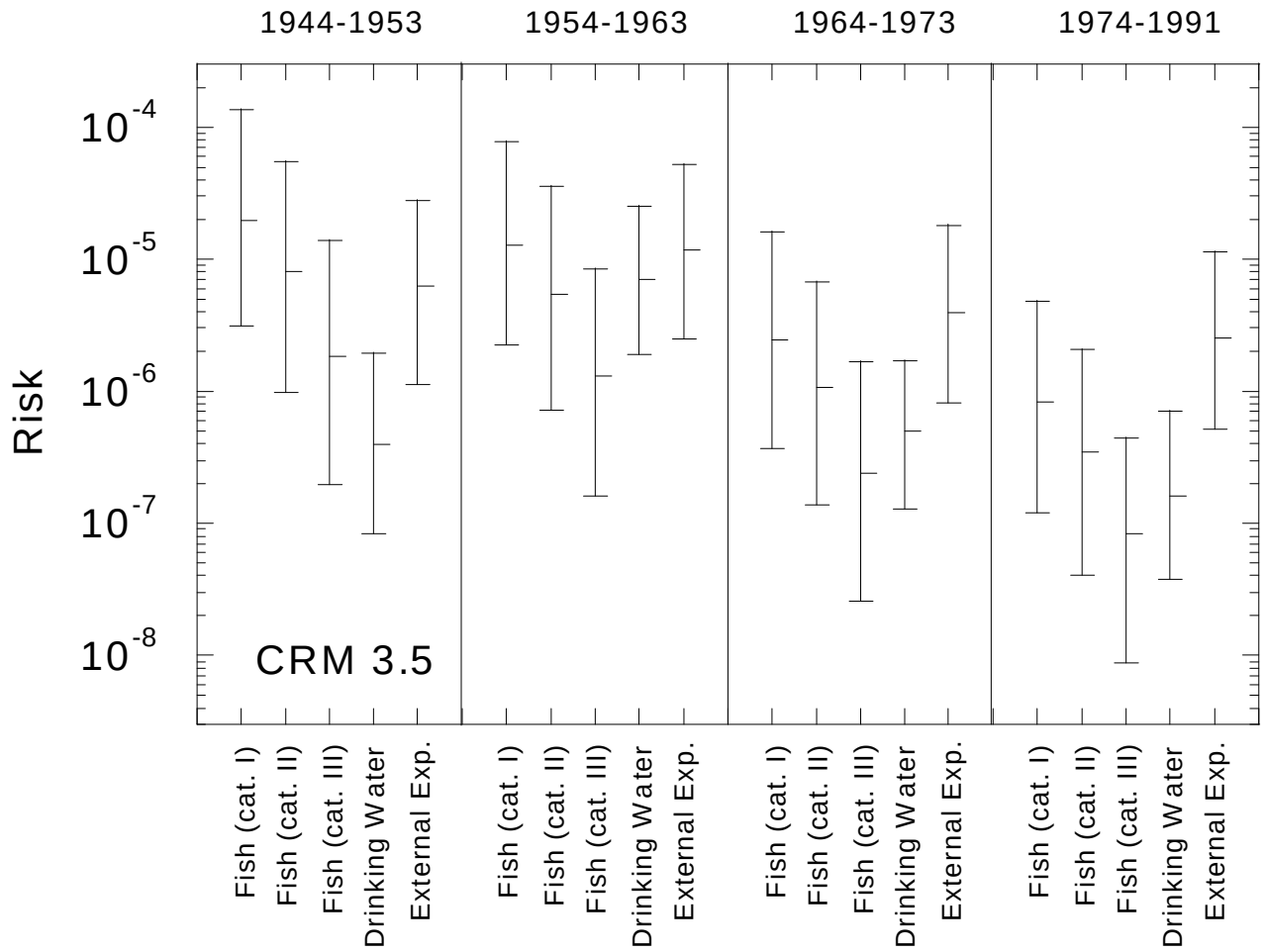

Figure 13.14 Estimates of total excess lifetime risk of cancer incidence for adult males, shown by exposure pathway at CRM 3.5 for four decades of interest. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; central values (50^{th} percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week.

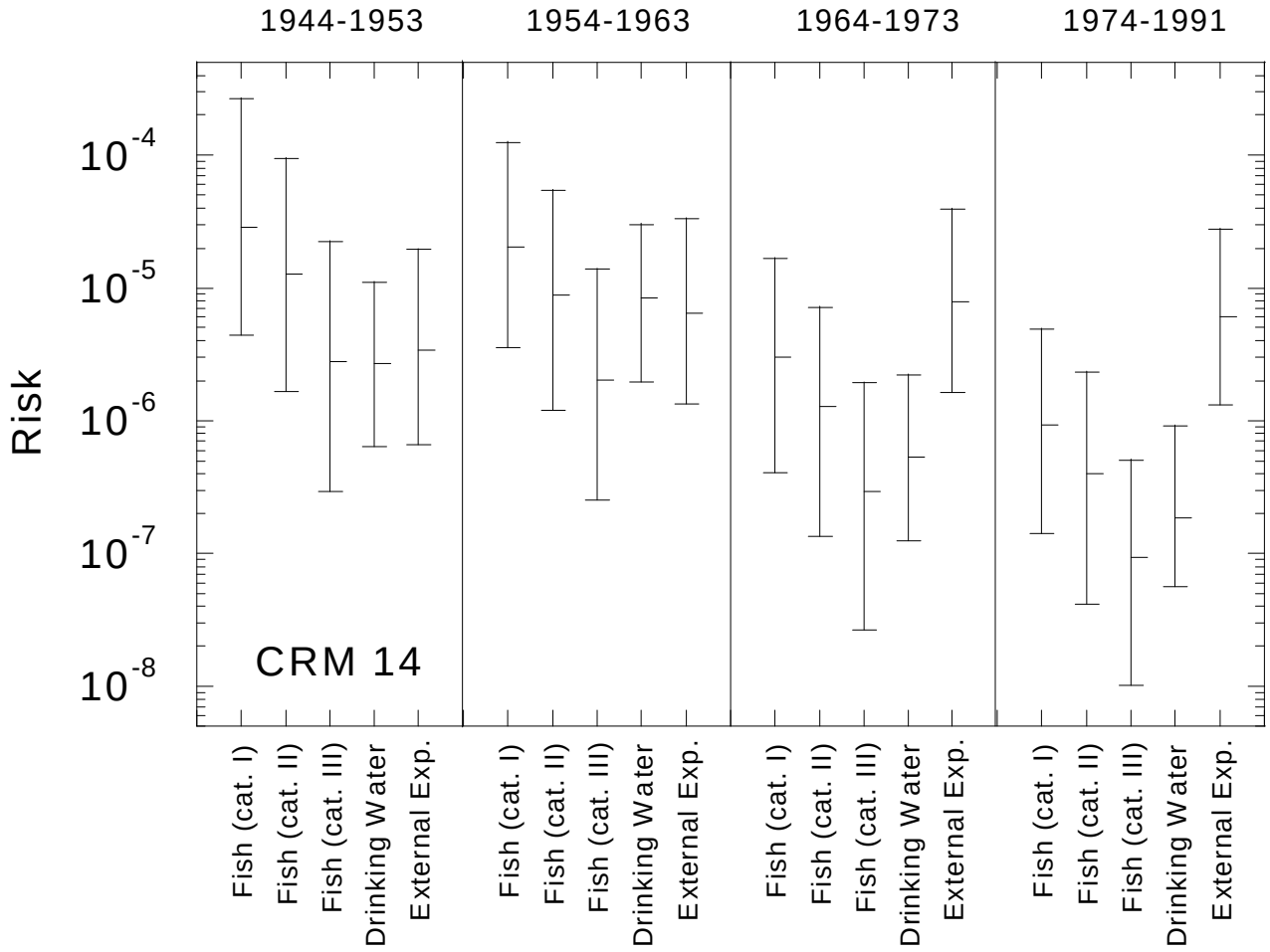

Figure 13.15 Estimates of total excess lifetime risk of cancer incidence for adult males, shown by exposure pathway at CRM 14 for four decades of interest. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; central values (50^{th} percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week.

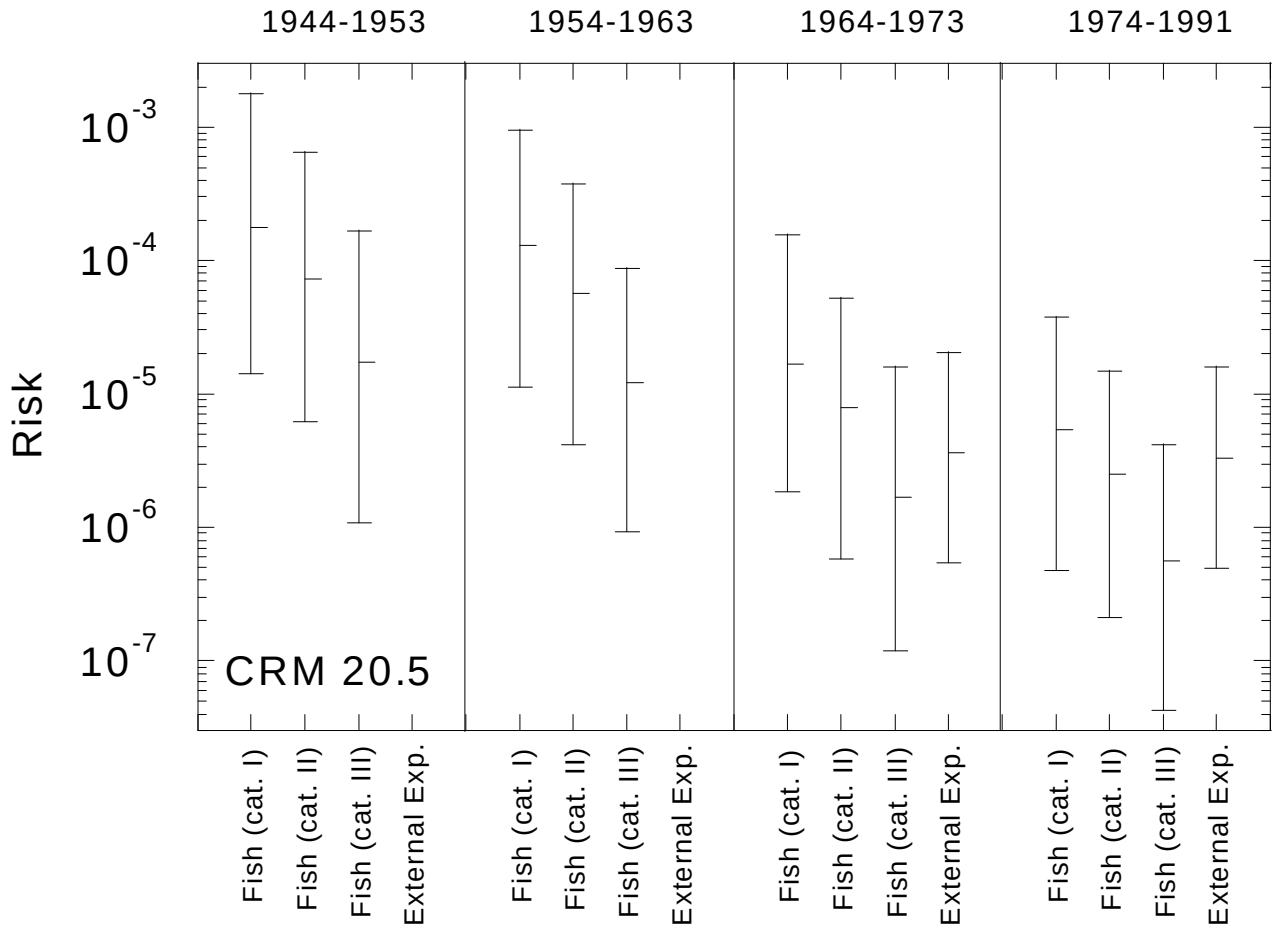

Figure 13.16 Estimates of total excess lifetime risk of cancer incidence for adult males, shown by exposure pathway at CRM 20.5 for four decades of interest. The vertical lines indicate the 95% subjective confidence intervals on the estimated risks; central values (50^{th} percentiles) are also indicated. Categories of fish consumption are as follows: Category I, 1-2.5 meals per week; Category II, 0.25-1.3 meals per week; Category III, 0.04-0.33 meals per week. The drinking water pathway did not occur at CRM 20.5.

13.5 Estimates of Annual Risk

Estimates of total risk of cancer incidence (all organs and radionuclides) from fish ingestion, external exposure, and drinking water were made for each year (1944-1991) in terms of the risk per pound (0.45 kg) of fish consumed (Table 13.11), per hour spent on the shoreline (Table 13.12), or per liter of water consumed (Table 13.13). All three tables show risk per unit exposure at CRM 14. External exposure risks at CRM 3.5 and CRM 0 are very similar to those at CRM 14. For fish ingestion, risks would be about a factor of 6 higher for CRM 20.5 than for CRM 14 or about a factor of 1.5 lower for CRM 3.5 and CRM 0 than for CRM 14. Unit risks for fish ingestion ranged from a lower bound of 7.6×10^{-11} in 1977 to an upper bound of 2.7×10^{-6} in 1946 (Table 13.11; these values assume that the fraction of contaminated fish was 1.0 and the fraction of radioactivity remaining after processing was 0.8 to 0.9). Thus, for example, a person who consumed 1 pound of Clinch River fish in 1946 would have an estimated risk of cancer incidence of 4.5×10^{-8} to 2.7×10^{-6}, while a person who consumed the same amount of fish in 1977 would have had a risk of only 8.7×10^{-11} to 1.0×10^{-9}. Annual unit risks from shoreline exposure ranged from a lower bound of 8.4×10^{-12} in 1944 to an upper bound of 2.9×10^{-8} in 1963 (Table 13.12). A person using the Clinch River shoreline for one hour in 1991 would have had a risk of 3.2×10^{-10} to 5.0×10^{-9}. For a person consuming one liter of water from the K-25 Water Intake, the unit risks of cancer incidence range from a lower bound of 1.3×10^{-12} in 1977 to an upper bound of 1.6×10^{-8} in 1960 (Table 13.13). Note that for estimation of risks for more than one year or pathway, the distributions must be propagated through the calculation (e.g., using Crystal Ball or similar software); they cannot simply be added arithmetically.

Table 13.11 Risks from the consumption of 1 pound (0.45 kg) of fish per year at CRM 14[a].

	95% subjective confidence interval		
Year	**lower bound**	**central value**	**upper bound**
1944	2.5×10^{-8}	2.2×10^{-7}	1.9×10^{-6}
1945	2.4×10^{-8}	1.8×10^{-7}	1.6×10^{-6}
1946	4.5×10^{-8}	3.3×10^{-7}	2.7×10^{-6}
1947	9.6×10^{-9}	9.3×10^{-8}	8.1×10^{-7}
1948	2.6×10^{-8}	1.7×10^{-7}	1.1×10^{-6}
1949	2.7×10^{-8}	1.2×10^{-7}	6.4×10^{-7}
1950	7.3×10^{-9}	3.4×10^{-8}	1.7×10^{-7}
1951	6.3×10^{-9}	2.8×10^{-8}	1.5×10^{-7}
1952	5.0×10^{-9}	2.1×10^{-8}	9.3×10^{-8}
1953	4.4×10^{-9}	1.9×10^{-8}	8.6×10^{-8}
1954	1.8×10^{-8}	6.7×10^{-8}	3.2×10^{-7}
1955	2.7×10^{-8}	1.1×10^{-7}	5.6×10^{-7}
1956	6.5×10^{-8}	2.9×10^{-7}	1.6×10^{-6}
1957	3.2×10^{-8}	1.4×10^{-7}	7.4×10^{-7}
1958	2.2×10^{-8}	8.8×10^{-8}	4.3×10^{-7}
1959	4.4×10^{-8}	2.0×10^{-7}	9.9×10^{-7}
1960	8.5×10^{-9}	3.0×10^{-8}	1.2×10^{-7}
1961	2.9×10^{-9}	1.1×10^{-8}	4.7×10^{-8}
1962	2.9×10^{-9}	1.0×10^{-8}	4.2×10^{-8}
1963	5.7×10^{-9}	2.3×10^{-8}	9.8×10^{-8}
1964	7.1×10^{-9}	2.9×10^{-8}	1.4×10^{-7}
1965	3.6×10^{-9}	1.5×10^{-8}	6.9×10^{-8}
1966	3.7×10^{-9}	1.4×10^{-8}	6.5×10^{-8}
1967	3.1×10^{-9}	1.3×10^{-8}	6.0×10^{-8}
1968	3.1×10^{-9}	1.3×10^{-8}	5.7×10^{-8}
1969	3.9×10^{-9}	1.7×10^{-8}	7.6×10^{-8}
1970	3.7×10^{-9}	1.6×10^{-8}	7.6×10^{-8}
1971	2.9×10^{-9}	1.0×10^{-8}	4.3×10^{-8}
1972	1.6×10^{-9}	5.6×10^{-9}	2.3×10^{-8}
1973	1.2×10^{-9}	4.0×10^{-9}	1.7×10^{-8}

Table 13.11 (continued)

	95% Subjective Confidence Interval		
Year	**lower bound**	**central value**	**upper bound**
1974	1.7×10^{-10}	5.5×10^{-10}	2.0×10^{-9}
1975	1.9×10^{-10}	6.6×10^{-10}	2.4×10^{-9}
1976	1.0×10^{-10}	3.3×10^{-10}	1.1×10^{-9}
1977	8.7×10^{-11}	2.9×10^{-10}	1.0×10^{-9}
1978	1.9×10^{-9}	8.8×10^{-9}	3.9×10^{-8}
1979	7.6×10^{-11}	2.7×10^{-10}	1.0×10^{-9}
1980	2.6×10^{-10}	8.4×10^{-10}	3.0×10^{-9}
1981	4.0×10^{-10}	1.3×10^{-9}	4.6×10^{-9}
1982	1.6×10^{-9}	5.5×10^{-9}	2.2×10^{-8}
1983	8.0×10^{-10}	2.7×10^{-9}	9.9×10^{-9}
1984	3.6×10^{-10}	1.4×10^{-9}	6.2×10^{-9}
1985	5.6×10^{-10}	1.8×10^{-9}	6.4×10^{-9}
1986	8.3×10^{-10}	3.3×10^{-9}	1.4×10^{-8}
1987	4.8×10^{-10}	2.0×10^{-9}	8.9×10^{-9}
1988	5.8×10^{-10}	2.3×10^{-9}	9.6×10^{-9}
1989	1.3×10^{-9}	4.4×10^{-9}	1.7×10^{-8}
1990	4.3×10^{-10}	1.5×10^{-9}	5.4×10^{-9}
1991	1.1×10^{-9}	4.8×10^{-9}	2.4×10^{-8}
Total[a]	4.5×10^{-7}	2.4×10^{-6}	1.5×10^{-5}

[a] To estimate risks for more than on year or pathway, the distributions must be propagated using Monte Carlo techniques; the values cannot simply be added.

Table 13.12 Risks from a 1-hour per year exposure to shoreline sediment at CRM 14[a].

	95% Subjective Confidence Interval		
Year	**lower bound**	**central value**	**upper bound**
1944	8.4×10^{-12}	6.6×10^{-11}	4.6×10^{-10}
1945	1.8×10^{-10}	1.2×10^{-9}	7.4×10^{-9}
1946	1.9×10^{-10}	1.8×10^{-9}	1.6×10^{-8}
1947	3.3×10^{-10}	2.0×10^{-9}	1.5×10^{-8}
1948	3.3×10^{-10}	2.4×10^{-9}	1.6×10^{-8}
1949	4.1×10^{-10}	2.0×10^{-9}	1.1×10^{-8}
1950	4.7×10^{-10}	2.5×10^{-9}	1.3×10^{-8}
1951	2.6×10^{-10}	1.2×10^{-9}	6.4×10^{-9}
1952	9.0×10^{-11}	4.3×10^{-10}	2.2×10^{-9}
1953	7.5×10^{-11}	4.6×10^{-10}	2.6×10^{-9}
1954	1.4×10^{-10}	6.8×10^{-10}	3.6×10^{-9}
1955	2.3×10^{-10}	1.1×10^{-9}	4.5×10^{-9}
1956	3.5×10^{-10}	1.4×10^{-9}	7.0×10^{-9}
1957	6.4×10^{-10}	2.7×10^{-9}	1.1×10^{-8}
1958	9.0×10^{-10}	3.7×10^{-9}	1.3×10^{-8}
1959	1.3×10^{-9}	4.9×10^{-9}	2.0×10^{-8}
1960	6.2×10^{-10}	2.6×10^{-9}	1.1×10^{-8}
1961	8.5×10^{-10}	3.2×10^{-9}	1.3×10^{-8}
1962	8.1×10^{-10}	3.3×10^{-9}	1.2×10^{-8}
1963	1.8×10^{-9}	7.7×10^{-9}	2.9×10^{-8}
1964	9.9×10^{-10}	4.2×10^{-9}	1.7×10^{-8}
1965	1.3×10^{-9}	5.7×10^{-9}	2.4×10^{-8}
1966	1.2×10^{-9}	4.7×10^{-9}	1.9×10^{-8}
1967	1.0×10^{-9}	4.6×10^{-9}	1.8×10^{-8}
1968	8.5×10^{-10}	3.6×10^{-9}	1.5×10^{-8}
1969	8.1×10^{-10}	3.3×10^{-9}	1.3×10^{-8}
1970	8.5×10^{-10}	4.2×10^{-9}	1.5×10^{-8}
1971	7.7×10^{-10}	3.3×10^{-9}	1.2×10^{-8}
1972	5.4×10^{-10}	2.4×10^{-9}	1.0×10^{-8}
1973	8.2×10^{-10}	3.3×10^{-9}	1.4×10^{-8}

Table 13.12 (continued)

	95% Subjective Confidence Interval		
Year	**lower bound**	**central value**	**upper bound**
1974	6.9×10^{-10}	2.9×10^{-9}	1.0×10^{-8}
1975	6.9×10^{-10}	3.0×10^{-9}	1.1×10^{-8}
1976	5.5×10^{-10}	2.3×10^{-9}	8.3×10^{-9}
1977	6.2×10^{-10}	2.5×10^{-9}	9.4×10^{-9}
1978	4.4×10^{-10}	2.0×10^{-9}	7.6×10^{-9}
1979	4.3×10^{-10}	1.7×10^{-9}	6.7×10^{-9}
1980	3.8×10^{-10}	1.6×10^{-9}	7.1×10^{-9}
1981	3.7×10^{-10}	1.6×10^{-9}	5.7×10^{-9}
1982	1.4×10^{-10}	5.8×10^{-10}	2.3×10^{-9}
1983	1.4×10^{-10}	6.2×10^{-10}	2.2×10^{-9}
1984	4.0×10^{-10}	1.7×10^{-9}	6.7×10^{-9}
1985	3.3×10^{-10}	1.4×10^{-9}	5.6×10^{-9}
1986	2.9×10^{-10}	1.3×10^{-9}	5.4×10^{-9}
1987	3.0×10^{-10}	1.3×10^{-9}	5.0×10^{-9}
1988	3.2×10^{-10}	1.3×10^{-9}	5.1×10^{-9}
1989	3.3×10^{-10}	1.4×10^{-9}	5.4×10^{-9}
1990	3.2×10^{-10}	1.3×10^{-9}	4.7×10^{-9}
1991	3.2×10^{-10}	1.3×10^{-9}	5.0×10^{-9}
Total[a]	2.6×10^{-8}	1.1×10^{-7}	4.9×10^{-7}

[a] To estimate risks for more than on year or pathway, the distributions must be propagated using Monte Carlo techniques; the values cannot simply be added.

Table 13.13 Risks from the consumption of 1 liter of drinking water per year at CRM 14[a].

	95% Subjective Confidence Interval		
Year	**lower bound**	**central value**	**upper bound**
1944	1.4×10^{-10}	5.9×10^{-10}	2.7×10^{-9}
1945	1.2×10^{-10}	5.2×10^{-10}	3.6×10^{-9}
1946	1.9×10^{-10}	8.9×10^{-10}	4.6×10^{-9}
1947	5.8×10^{-11}	2.3×10^{-10}	1.5×10^{-9}
1948	1.0×10^{-10}	4.3×10^{-10}	2.0×10^{-9}
1949	1.9×10^{-10}	7.1×10^{-10}	3.2×10^{-9}
1950	3.8×10^{-11}	1.4×10^{-10}	6.2×10^{-10}
1951	3.2×10^{-11}	1.1×10^{-10}	5.2×10^{-10}
1952	7.6×10^{-11}	2.7×10^{-10}	1.4×10^{-9}
1953	1.3×10^{-10}	5.4×10^{-10}	2.5×10^{-9}
1954	2.1×10^{-10}	7.8×10^{-10}	4.0×10^{-9}
1955	1.1×10^{-10}	4.3×10^{-10}	2.0×10^{-9}
1956	1.4×10^{-10}	4.8×10^{-10}	2.3×10^{-9}
1957	1.1×10^{-10}	4.0×10^{-10}	1.8×10^{-9}
1958	1.7×10^{-10}	5.9×10^{-10}	2.8×10^{-9}
1959	5.6×10^{-10}	2.1×10^{-9}	9.9×10^{-9}
1960	9.7×10^{-10}	3.7×10^{-9}	1.6×10^{-8}
1961	7.2×10^{-10}	2.8×10^{-9}	1.2×10^{-8}
1962	4.2×10^{-10}	1.6×10^{-9}	6.9×10^{-9}
1963	2.3×10^{-10}	8.7×10^{-10}	3.6×10^{-9}
1964	1.3×10^{-10}	5.0×10^{-10}	2.0×10^{-9}
1965	4.2×10^{-11}	1.5×10^{-10}	5.8×10^{-10}
1966	2.2×10^{-11}	7.8×10^{-11}	2.9×10^{-10}
1967	8.0×10^{-12}	2.7×10^{-11}	1.1×10^{-10}
1968	9.4×10^{-12}	3.3×10^{-11}	1.4×10^{-10}
1969	1.0×10^{-11}	3.5×10^{-11}	1.3×10^{-10}
1970	7.5×10^{-12}	2.6×10^{-11}	1.0×10^{-10}
1971	1.7×10^{-11}	5.2×10^{-11}	1.9×10^{-10}
1972	9.3×10^{-12}	3.1×10^{-11}	1.2×10^{-10}
1973	6.9×10^{-12}	2.3×10^{-11}	8.9×10^{-11}

Table 13.13 (continued)

	95% Subjective Confidence Interval		
Year	**lower bound**	**central value**	**upper bound**
1974	2.7×10^{-12}	9.5×10^{-12}	3.7×10^{-11}
1975	2.1×10^{-12}	7.3×10^{-12}	2.8×10^{-11}
1976	1.6×10^{-12}	5.3×10^{-12}	2.1×10^{-11}
1977	1.3×10^{-12}	4.1×10^{-12}	1.5×10^{-11}
1978	3.7×10^{-12}	1.0×10^{-11}	3.8×10^{-11}
1979	1.8×10^{-12}	6.4×10^{-12}	2.5×10^{-11}
1980	3.5×10^{-12}	1.2×10^{-11}	4.8×10^{-11}
1981	5.6×10^{-12}	2.1×10^{-11}	8.3×10^{-11}
1982	7.2×10^{-12}	2.6×10^{-11}	1.0×10^{-10}
1983	7.8×10^{-12}	2.9×10^{-11}	1.2×10^{-10}
1984	4.4×10^{-12}	1.6×10^{-11}	6.3×10^{-11}
1985	7.2×10^{-12}	2.6×10^{-11}	1.1×10^{-10}
1986	7.3×10^{-12}	2.7×10^{-11}	1.1×10^{-10}
1987	1.2×10^{-11}	4.6×10^{-11}	1.9×10^{-10}
1988	7.2×10^{-12}	2.7×10^{-11}	1.1×10^{-10}
1989	8.4×10^{-12}	3.1×10^{-11}	1.3×10^{-10}
1990	3.2×10^{-12}	1.2×10^{-11}	4.9×10^{-11}
1991	3.7×10^{-12}	1.4×10^{-11}	7.3×10^{-11}
Total[a]	5.1×10^{-9}	2.0×10^{-8}	8.9×10^{-8}

[a] To estimate risks for more than on year or pathway, the distributions must be propagated using Monte Carlo techniques; the values cannot simp ly be added.

13.6 Comparison to Federal Guidance 13

Risks per unit intake for ^{137}Cs, ^{106}Ru, ^{90}Sr, and ^{60}Co estimated in this analysis were compared with the standard risk factors presented in Federal Guidance 13 (Figure 13.17; Eckerman et al., 1998). In general, the central values obtained in this analysis were similar to or slightly lower than the values obtained with Federal Guidance 13; for ^{60}Co, the Federal Guidance 13 value was substantially higher. The exact reasons for this difference are unknown at present, but part of the bias is probably due to the higher GI tract absorption assumed in the Federal Guidance 13 model than was assumed in the present analysis. Thus, the Federal Guidance 13 estimates, although intended to produce best-estimate results, may still contain some amount of conservative bias (especially for ^{60}Co). The results produced in this analysis have included the uncertainties in both the dosimetry and the dose-to-risk estimation. Federal Guidance 13 currently does not include the results of an uncertainty analysis.

13.7 Identification of the Important Contributors to the Uncertainty in the Organ-Specific Cancer Risk Estimates

An initial sensitivity analysis was performed to assess the contributions to the uncertainty in the estimates of total excess lifetime risk for each exposure pathway from each major component of the assessment. Sensitivity analyses were carried out for male Category I consumers of fish at CRM 20.5 and CRM 14, for male Category III consumers at CRM 14, and for external exposure and ingestion of drinking water at CRM 14.

Because ^{137}Cs was the dominant contributor to dose and risk from fish ingestion at all consumption rates, the sensitivity analyses for fish ingestion were carried out only for the risk from ^{137}Cs. For all ingestion rates and locations examined, the radionuclide concentration in fish and the amount of fish consumed were the dominant contributors to the uncertainty in risk (Figures 13.18 to 13.20). Lesser contributors included the radionuclide concentration in water, the dose-response, and the dosimetry. For the Category I consumer at CRM 20.5, more than 50% of the uncertainty is attributed to the radionuclide concentration in fish; this in turn is due both to the uncertainty in the bioconcentration factor and to the additional adjustment factor used for CRM 20.5 to account for fish swimming in and out of White Oak Creek Embayment or the plume from White Oak Creek (Section 8). For CRM 14, uncertainty in the radionuclide concentration in fish was due only to the bioconcentration factor; the bioconcentration factor accounted for about one-third and one-fourth of the uncertainty in risk for Category I and Category III consumers at CRM 14, respectively. Uncertainty in the amount of fish consumed accounted for about one-fourth of the uncertainty in risk for the Category I consumer at CRM 20.5, one-third for the Category I consumer at CRM 14, and almost half for the Category III consumer at CRM 14.

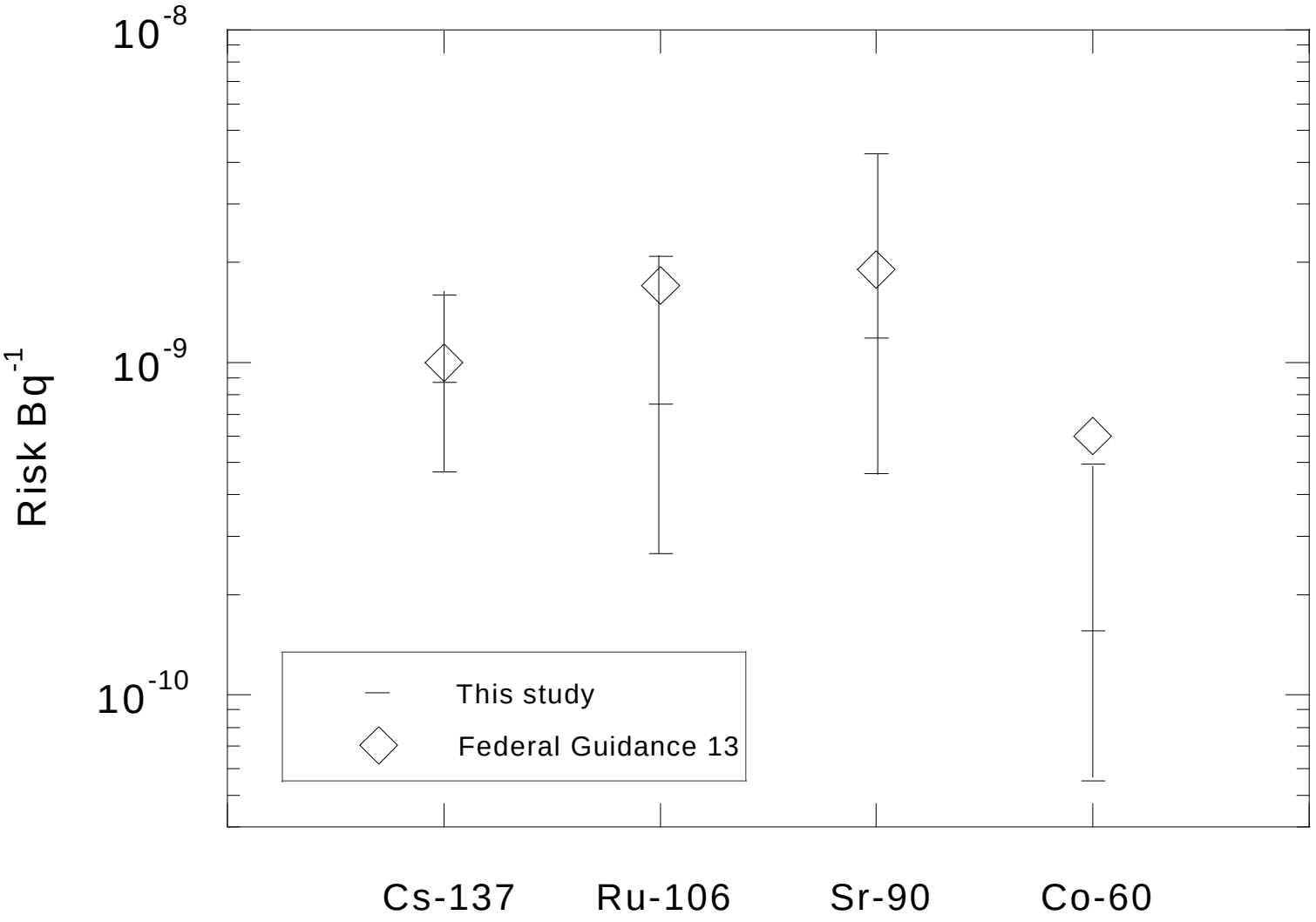

Figure 13.17 Comparison of risk per unit intake from this analysis with published risk factors in Federal Guidance 13 (Eckerman et al., 1998). The vertical lines indicate the 95% subjective confidence intervals on the risk per unit intake used in the present study; central values (50th percentiles) are indicated by small horizontal lines. Open diamonds indicate the values from Federal Guidance 13.

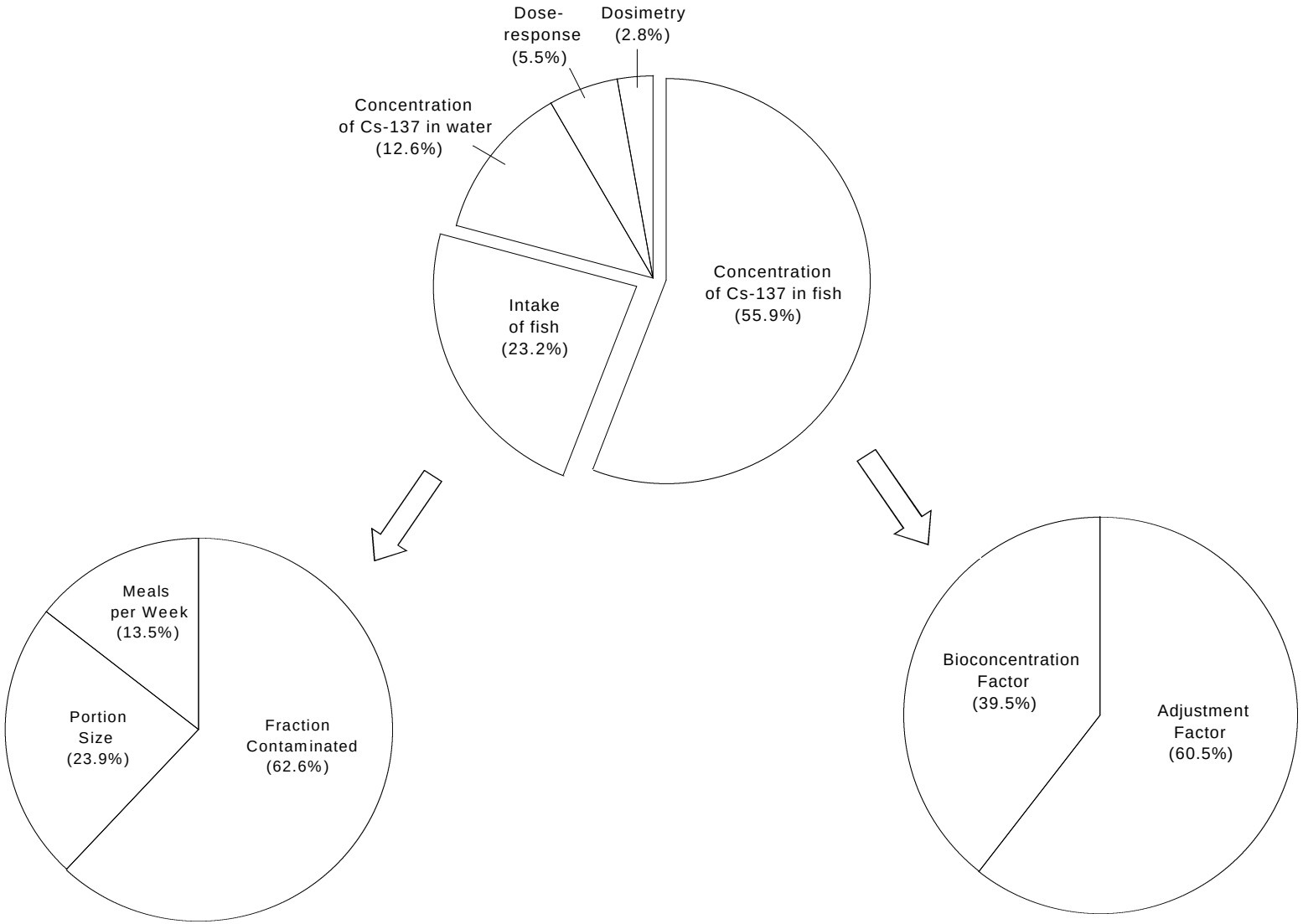

Figure 13.18 Results of a sensitivity analysis for the risk from ^{137}Cs to male Category I consumers of fish (1-2.5 meals per week) at CRM 20.5. The graphs show the contribution of each uncertain input to the total uncertainty in the estimate of excess lifetime risk.

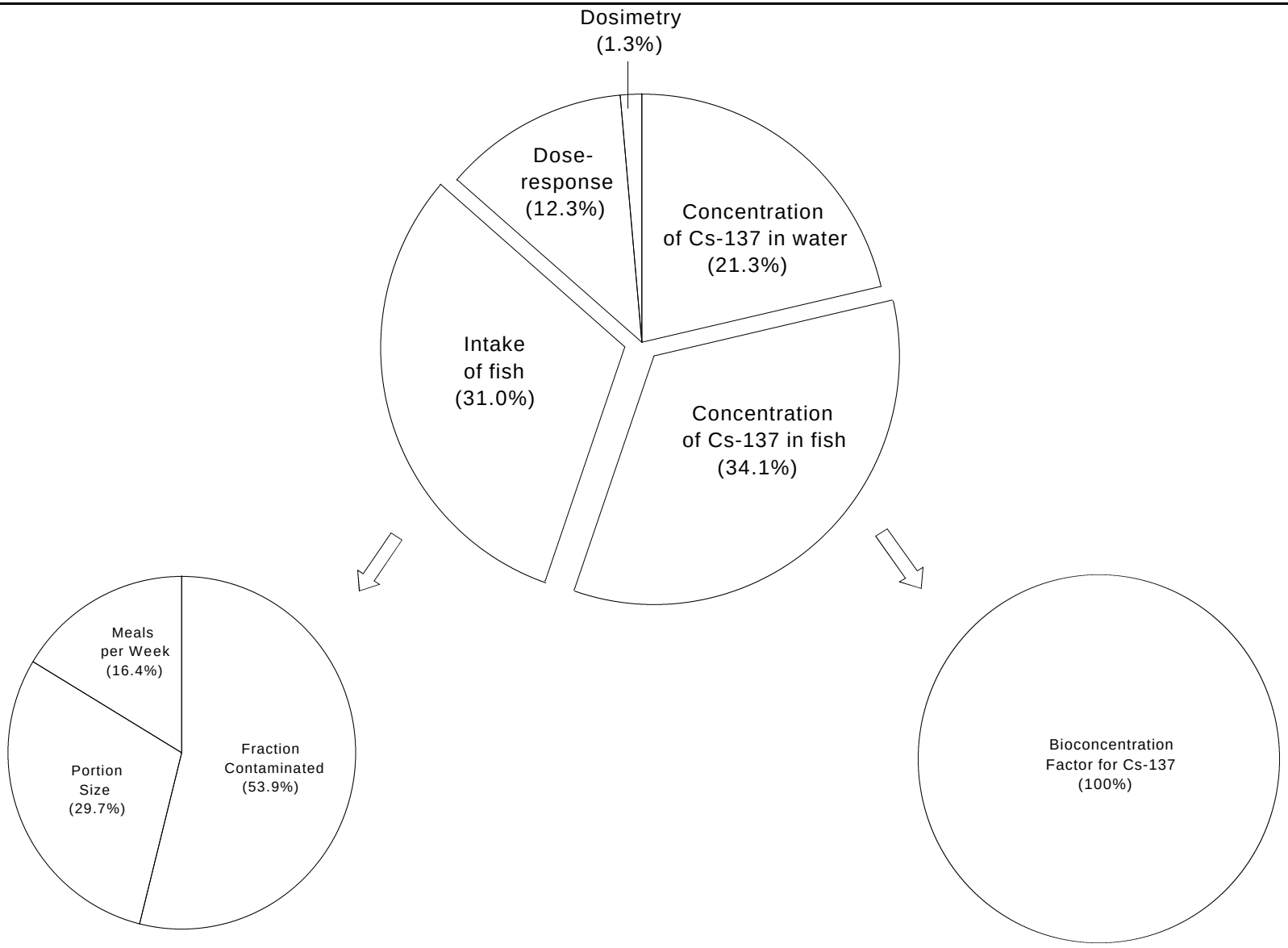

Figure 13.19 Results of a sensitivity analysis for the risk from ^{137}Cs to male Category I consumers of fish (1-2.5 meals per week) at CRM 14. The graphs show the contribution of each uncertain input to the total uncertainty in the estimate of excess lifetime risk.

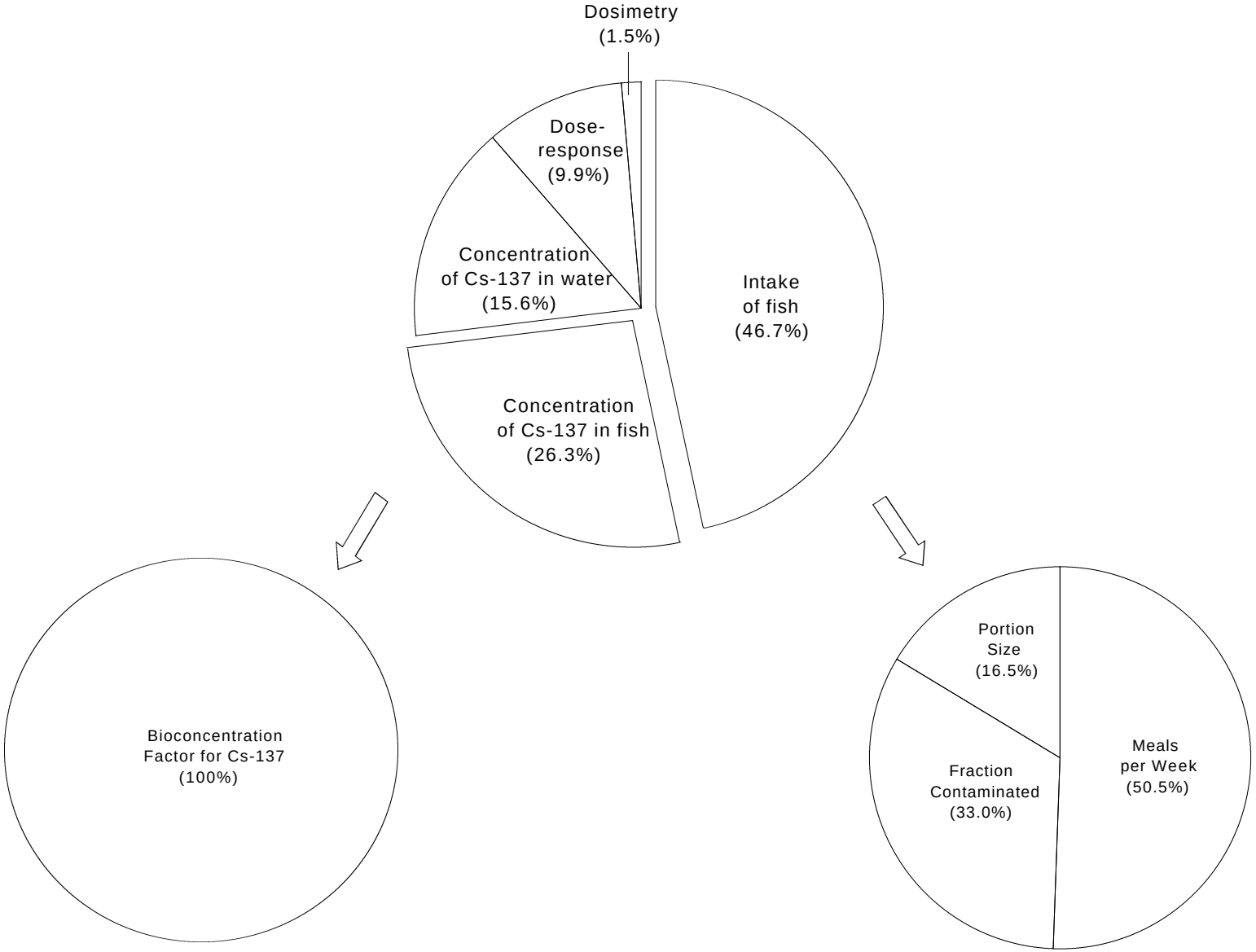

Figure 13.20 Results of a sensitivity analysis for the risk from ^{137}Cs to male Category III consumers of fish (0.04-0.33 meals per week) at CRM 14. The graphs show the contribution of each uncertain input to the total uncertainty in the estimate of excess lifetime risk.

For external exposure at CRM 14, the dominant contributors to risk are ^{137}Cs and ^{60}Co. For both radionuclides, the largest source of uncertainty was the predicted radionuclide concentration in sediment (Figures 13.21). The dose-response (risk factor) and factors affecting the amount of exposure (exposure time, shoreline width, sediment depth, and position on shoreline) were also important contributors to the overall uncertainty in risk.

The dominant contributors to risk from drinking water at CRM 14 are ^{106}Ru and ^{90}Sr. For both radionuclides, uncertainties in the total intake (primarily the uncertainty in the intake rate of water and the fraction of water from the contaminated source), contribute nearly half of the uncertainty in the overall risk from drinking water, while the radionuclide concentration in water and the dose-response (risk factor) contribute most of the rest (Figure 13.22).

Thus for internal exposure to ^{137}Cs (via ingestion of fish), the amount of contaminated food (fish) consumed and the concentration of ^{137}Cs in the food are the dominant sources of uncertainty. For internal exposure to ^{90}Sr and ^{106}Ru (via drinking water), the amount consumed dominates the uncertainty, but the risk factor is comparable to or greater than the radionuclide concentration in water in terms of importance to the overall uncertainty in the risk. For external exposure to ^{137}Cs and ^{60}Co, the risk factor and the exposure situation are of similar importance behind the radionuclide concentration in the shoreline sediment. Uncertainty in dosimetry contributes less than 5% (internal) or 10% (external) of the uncertainty, while the risk factor contributes 20-30% (except for internal exposure to ^{137}Cs, for which the risk factor contributes about 10% of the uncertainty). Uncertainties in exposure parameters (radionuclide concentrations and amounts of exposure) are dominant for all pathways.

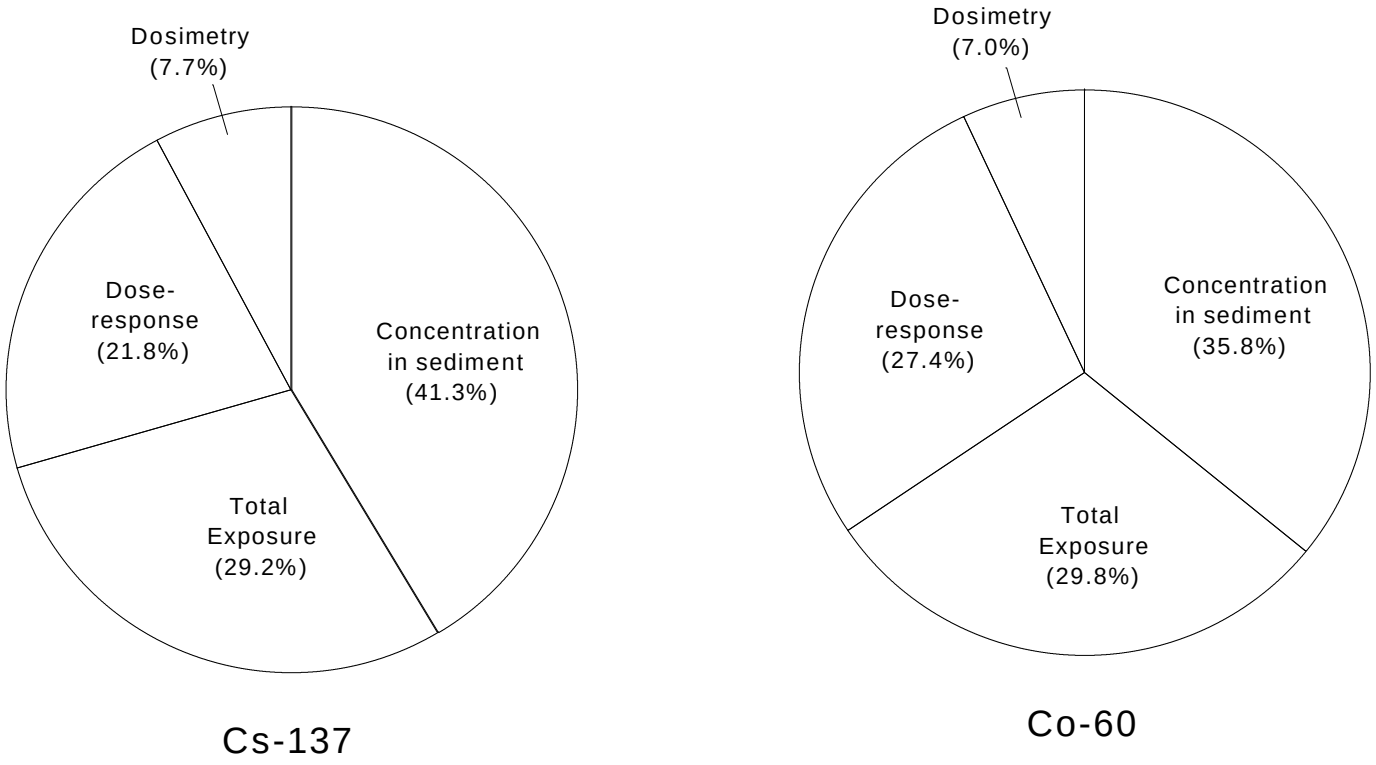

Figure 13.21 Results of a sensitivity analysis for the risk from external exposure to ^{137}Cs and ^{60}Co in shoreline sediments at CRM 14. The graphs show the contribution of each uncertain input to the total uncertainty in the estimate of excess lifetime risk.

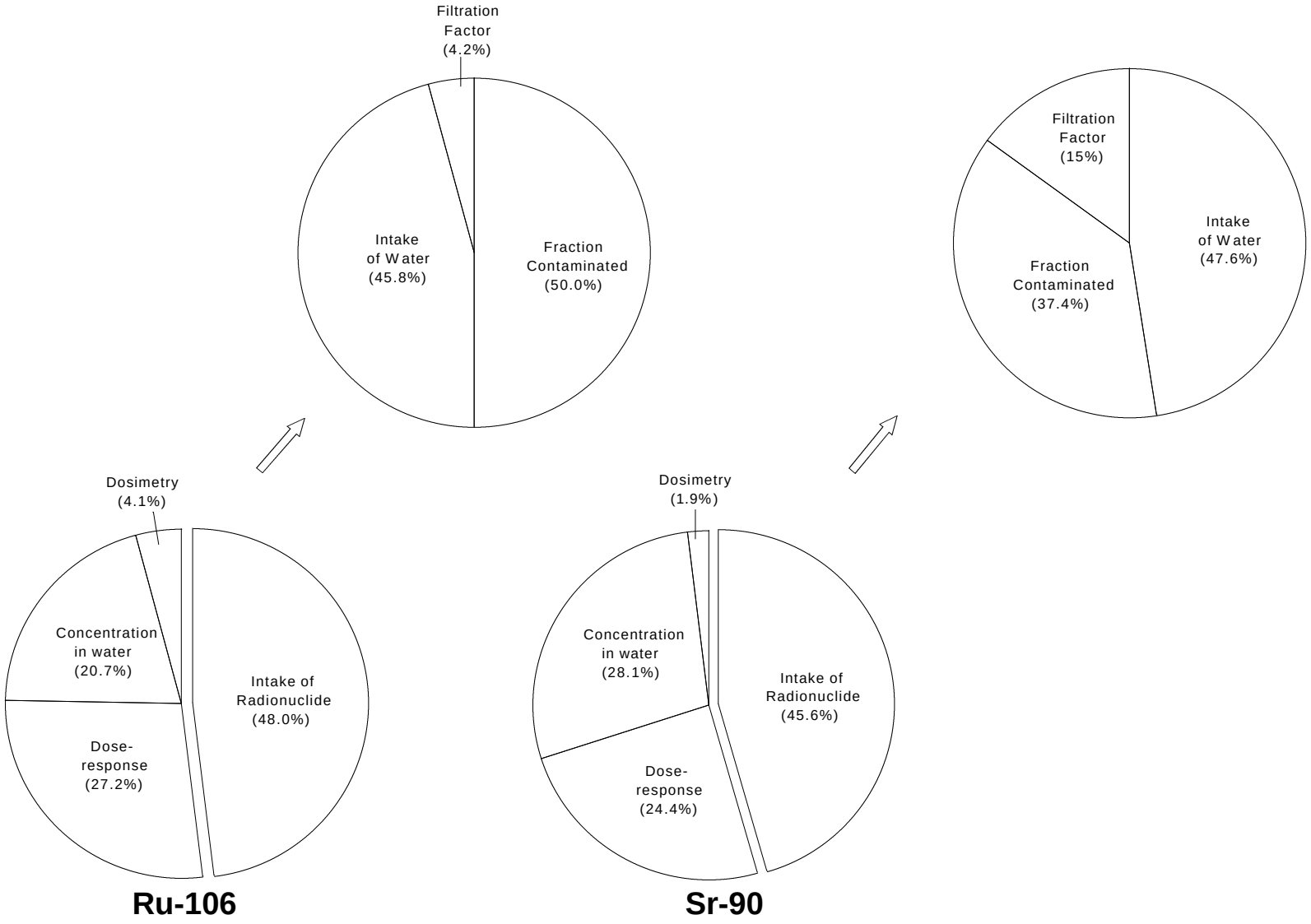

Figure 13.22 Results of a sensitivity analysis for the risk from ^{90}Sr and ^{106}Ru in drinking water at CRM 14. The graphs show the contribution of each uncertain input to the total uncertainty in the estimate of excess lifetime risk.

13.8 What Do the Results Mean for Individuals Fishing in Watts Bar Reservoir?

While radionuclide concentrations in Clinch River fish provide an upper bound on concentrations expected in fish in the Tennessee River or Watts Bar Reservoir, average concentrations in those fish tend to be as much as a factor of 25 lower than in Clinch River fish (depending on the radionuclide), due to the greater dilution afforded by the Tennessee River (Section 8). Consumption patterns for the Tennessee River/Watts Bar Reservoir fish may differ from those for the Clinch River, and more people are likely to have eaten fish from the Tennessee River/Watts Bar Reservoir than from the Clinch River. However, it is unlikely that anyone would have consistently eaten more than the Category I consumer defined in this analysis over the entire period of 48 years.

The 95% subjective confidence limit for the excess lifetime risk of cancer incidence for a Category I consumer (1-2.5 meals per week) in the K-25/Grassy Creek area (CRM 14) is 8.2×10^{-6} to 4.1×10^{-4} for males and 6.7×10^{-6} to 3.4×10^{-4} for females; most of this risk is contributed by ^{137}Cs. Concentrations of ^{137}Cs in the Tennessee River/Watts Bar Reservoir have historically been a factor of 4 to 25 lower than at CRM 14, hence a person with the same ingestion pattern as the Category I consumer in this analysis would be expected to have an excess lifetime risk of cancer incidence of about a factor of 4 to 25 less than a person with the same consumption rate at CRM 14.

13.9 Comparison with Other Dose Reconstruction Studies

13.9.1 Comparison with the Cowser and Snyder (1966) Safety Analysis

Cowser and Snyder (1966) performed a safety analysis of radionuclides released from the X-10 site to the Clinch River; this analysis included an evaluation of internal doses to selected organs and to the whole body. The radiation dose estimates were based on estimated concentrations of radionuclides in the Clinch and Tennessee Rivers from 1944-1963. The major radionuclides of concern were ^{137}Cs, ^{90}Sr, ^{106}Ru (especially 1959-63), and ^{131}I, although ^{60}Co and several others were also included.

Radiation doses were assessed on an annual basis in terms of the fraction of "Maximum Permissible Concentrations" [MPC, based on ICRP Publication 2 (1959)] of radionuclides in water. Most values for MPC were derived from a value of 0.015 Sv (1.5 rem) per year for an individual and 0.005 Sv (0.5 rem) per year for averages of exposed populations, based on 1/10 of the MPC values for occupational exposure. The maximum annual fractions of MPC (based on annual average radionuclide concentrations) for CRM 14.5 were 0.13 for bone (1954), 0.050 for the gastrointestinal tract (1960), 0.072 for the total body (1954), and 0.022 for the thyroid (1954).

Estimated cumulative doses from 1944-1963 from three pathways (drinking water, recreational [external] exposure, and fish consumption) were 3.2 cSv (3.2 rem) for the skeleton and 0.27 cSv (0.27 rem) for the total body for the Clinch River and 0.45 cSv (0.45 rem) for the skeleton and 0.039 cSv (0.039 rem) for the total body for the Tennessee River. These early dose estimates were thought to be conservative. Dose estimates for the Clinch River were based on an 18-year-old male employed at the K-25 site (one-half of daily fluid intake occurring at the job site); dose estimates for the Tennessee

River were based on a 14-year-old male obtaining water from a municipal supply on the Tennessee River. For fish ingestion, dose estimates for the Clinch were based on consumption of 10.9 kg (24 lb) of Clinch River fish per year; estimates for the Tennessee assumed dilution of the supply with fish from other sources. Recreational (external) exposure was a relatively insignificant contributor to the estimated total doses.

To put the Cowser and Snyder study in context with the present analysis, a total dose of 3.2 cSv is equivalent to a total risk of cancer incidence of about 2×10^{-3} (based on 0.06 risk per Sv). This is very close to the upper bound of the total risk estimate for Category I and II consumers near Jones Island (CRM 20.5) in this analysis. However, in the present analysis, most of the estimated risk is from fish ingestion, while in Cowser and Snyder's analysis, about half was from fish ingestion (56%) and half (44%) from drinking water. The fish consumption rate used by Cowser and Snyder was between the 50^{th} and 95^{th} percentiles of the rate (or range of rates) used for a Category I consumer in the present analysis, and all of it (vs. 20 to 100% in this study) was assumed to be contaminated.

13.9.2 Comparison with the Columbia River Dose Reconstruction

A comparison was made between the results of the present study and results from the Columbia River Dose Reconstruction for the Hanford (Washington) Site (Farris et al., 1994; Hoffman et al., 1998). The Columbia River Dose Reconstruction was primarily concerned with releases of ^{32}P, ^{65}Zn, ^{24}Na, ^{76}As, and ^{239}Np. The study assessed doses and risks to a defined "maximum representative" individual (MRI) exposed over 20 years near Richland, Washington (1950 to 1970, approximately 20 miles from the release point). The maximum representative individual was assumed to eat 40 kg y^{-1} of resident fish and 20 kg y^{-1} of waterfowl. As described in Sections 7 and 13.1 of the present report, the Category I consumer in this study was assumed to eat 1-2.5 meals per week of fish, of which 20-100% was assumed to be contaminated; 80-90% of the contamination was assumed to remain after processing. For males (meal size = 0.10-0.30 kg), the total consumption of fish amounts to 7.1 – 33 kg y^{-1} (95% subjective confidence interval) and for females (meal size = 0.08 – 0.25 kg), 5.7 – 27 kg y^{-1} (95% subjective confidence interval). Thus the ingestion rate for the Columbia River MRI exceeds the 97.5^{th} percentile of the ingestion rate for Category I consumers in the present study, before the application (for the Clinch River consumer) of the adjustments for fraction contaminated and amount remaining after processing. Waterfowl were included in the dose and risk estimates for the Columbia River but not for the Clinch River; the exposure from the Clinch River occurred over 48 years, as opposed to 20 years for the Columbia River.

This comparison indicates that for specific organs the range of excess lifetime risk of cancer incidence for male and female adults who are Category I consumers of fish is generally lower than the risks from fish ingestion for the "maximum representative" individual defined for the Columbia River study (Figure 13.23). In the Columbia River Study, the organs of interest are the colon and red bone marrow (risk primarily from ^{32}P) and the breast (risk primarily from ^{65}Zn). The risks for leukemia and colon cancer were compared for the Columbia River Study and for male Category I consumers of fish near the Jones Island Area and the K-25/Grassy Creek area of the Clinch River (Figure 13.23); the risks for the breast

for females in the Oak Ridge Study were also compared with the "maximum representative individual" in the Columbia River Study. In general, the upper half of the range for the CRM 20.5 individual overlaps the lower half of the range (nearly all of the range for breast) predicted for the Columbia River study. The difference in the risks at the two sites reflect the combined differences in ingestion rates, contamination levels, and exposure durations for the two situations.

13.10 Influence of Radionuclides from Weapons Testing Fallout

An additional source of ^{90}Sr and ^{137}Cs exposure for the residents of Tennessee may have been fallout from the atmospheric testing of nuclear weapons. Evaluation of measured water concentrations both upstream and downstream of White Oak Creek showed comparable levels for the two areas for several years (Section 6). In addition, Cowser and Snyder (1966) estimated that 45% of the ^{90}Sr and 20% of the ^{137}Cs in the Clinch River in 1962-63 was from weapons testing fallout, and Churchill et al. (1965) reported that higher levels of ^{90}Sr were observed at Chattanooga (downstream on the Tennessee River) than could be accounted for by releases from White Oak Dam. Because measured concentrations of radionuclides in water were used in this study when available, including much of the period during which weapons testing occurred (Section 6), the calculations of dose and risk in this study include the effects of fallout ^{90}Sr and ^{137}Cs (except for external exposure calculations which were based solely on modeled concentrations of radionuclides in sediment). The weapons testing fallout may well have been a significant contributor to the total ^{90}Sr and ^{137}Cs exposures in some years, but fallout would not have contributed significantly to the total excess lifetime risks of cancer incidence for the entire period of historical releases. These risks are dominated by the early years of exposure (pre-1960), when the White Oak Creek releases were the highest.

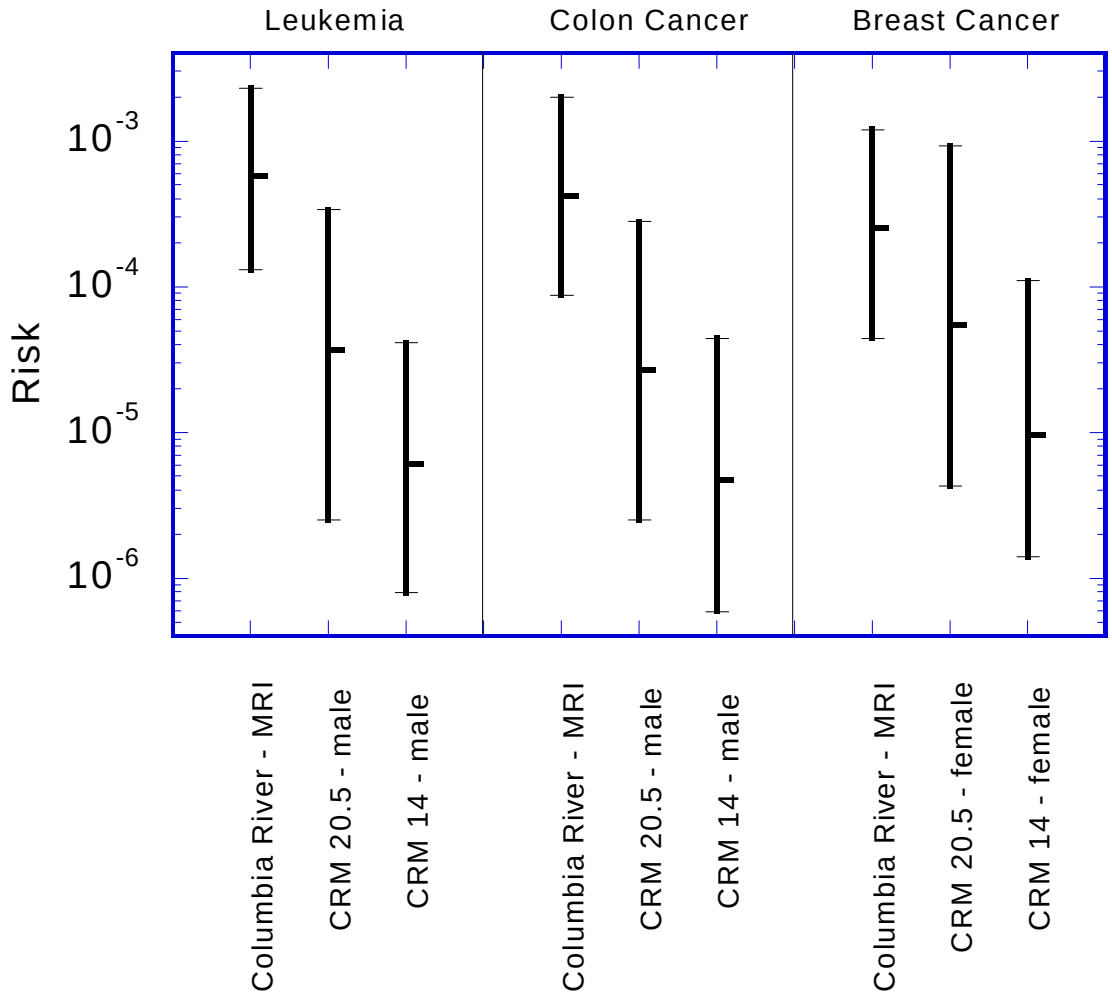

Figure 13.23 Comparison of risk estimates for selected cancer types from ingestion of contaminated fish from the Columbia River (WA and OR) and the Clinch River (TN). The exposed individuals are defined as "maximum representative individual" (MRI; 40 kg y^{-1} of fish and 20 kg y^{-1} of waterfowl, all of which is contaminated) for the Columbia River, and as "Category I" (7.1-33 kg y^{-1} for males or 5.7-27 kg y^{-1} for females, of which 20-100% is contaminated) for the Clinch River. The exposure locations are Richland on the Columbia River and CRM 20.5 and CRM 14 for the Clinch River.

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14.0 ESTIMATES OF EXCESS LIFETIME RISK OF CANCER INCIDENCE FOR SPECIAL SCENARIOS

In addition to the major pathways of exposure addressed in Section 13, two types of special scenarios for internal exposure to radionuclides from the X-10 site have been examined. These include the ingestion of bones as well as flesh of Clinch River fish, in the form of fish patties made by grinding small fish (Section 14.1), and ingestion of contaminated animals (fish, deer, waterfowl, or turtles) that migrated off the Oak Ridge Reservation (Section 14.2). These two special cases are not likely to have affected very many individuals, but such cases are known or suspected to have occurred.

14.1 Consumption of Fish Patties

Anecdotal evidence suggests that some people made fish patties from small fish obtained from the Clinch River (ORNL, 1985). The fish were cleaned (scales, head, and internal organs removed), and the remaining parts of the fish, including flesh and bones, were put into a meat grinder. This scenario deals only with the fish portion of the patties and omits consideration of other ingredients that might have been added (e.g., bread crumbs). The primary concern with respect to consumption of fish patties is the likelihood of increased doses due to ^{90}Sr that had accumulated in the bones of the fish. Strontium-90 is known to concentrate in bone and therefore would be present in higher concentrations in fish patties than in fish flesh.

For this scenario, it was assumed that the diet of an avid consumer of fish (Category I, or approximately 7.1 to 33 kg y^{-1} for males and 5.7-27 kg y^{-1} for females; see Section 7) consisted of a fish-patty meal at least once a month but no more than twice a month (8 to 20% of total fish meals in the form of fish patties). An adjustment factor was applied to the fish-patty portion of the ingested fish to account for the additional ^{90}Sr in the patty due to the inclusion of bones. The fish were assumed to have been taken from CRM 14; the relationship of the doses and risks upstream or downstream of CRM 14 would be the same as those seen for the previous estimates for ingestion of fish (about a factor of 6 higher at CRM 20.5 or a factor of 1.5 lower at CRM 0 and 3.5; Section 13).

Selected organ-specific doses for male and female Category I consumers of fish, with and without fish patties, are presented in Table 14.1. For both males and females, the highest doses in both cases (with and without fish patties) were to bone and red bone marrow. Because of the increased contribution of ^{90}Sr to the doses in the fish patty scenario, doses to the bone and red bone marrow of the target individuals are 15-25% higher with fish patties than with fish flesh only; doses to other organs are about the same.

Table 14.1 Selected organ-specific doses (cSv) from all radionuclides at CRM 14 for Category I consumers[a] of fish with and without fish patties.

Organ	Fish flesh alone	Fish flesh and fish patties[b]
Male		
Bone	0.014-0.90 (0.13)[c]	0.022-1.2 (0.15)
Lower large intestine	0.017-0.64 (0.095)	0.018-0.69 (0.093)
Red bone marrow	0.015-0.64 (0.096)	0.018-0.70 (0.11)
Skin	0.0079-0.35 (0.054)	0.0080-0.38 (0.051)
Female		
Bone	0.016-0.81 (0.11)	0.016-0.99 (0.13)
Breast	0.0068-0.35 (0.046)	0.0070-0.31 (0.044)
Lower large intestine	0.014-0.46 (0.079)	0.014-0.54 (0.080)
Red bone marrow	0.015-0.49 (0.075)	0.014-0.63 (0.093)
Skin	0.0065-0.29 (0.045)	0.0070-0.30 (0.043)

[a] The fish consumption rate for Category I individuals is defined by a 95% subjective confidence interval of 7.1-33 kg y^{-1} for males (central value, 17 kg y^{-1}) and 5.7-27 kg y^{-1} for females (central value, 14 kg y^{-1}) See Sections 7 and 13 for more details.

[b] Between 8 and 20% of the total fish consumed is in the form of fish patties.

[c] The range indicates the 95% subjective confidence interval, and the value in parentheses indicates the central value (50th percentile).

The excess lifetime risks of cancer incidence for male and female Category I consumers, with and without fish patties, are shown in Table 14.2. The risks from individual radionuclides and from total radionuclide contaminants are given for fish flesh alone and for the combination of fish flesh and fish patties (same total consumption, but 8-20% in the form of patties). Although the risk from ^{90}Sr increases about 40% (based on the 50th percentiles) to 100% (based on the 97.5th percentiles) for people consuming fish patties as well as fish flesh, the total risk is essentially the same. This situation is due to the fact that the primary doses and risks from fish consumption are a result of ^{137}Cs contamination. While the doses and risks to the bone and bone marrow are increased by the consumption of additional ^{90}Sr, these doses and risks are small in comparison to the total doses and risks from ^{137}Cs and thus do not contribute significantly to the total doses and risks from consumption of fish.

Table 14.2 Excess lifetime risk of cancer incidence for Category I consumers[a] of fish at CRM 14, with and without fish patties.

Organ	Fish flesh alone	Fish flesh and fish patties[b]
Male		
Risk from ^{137}Cs	6.7×10^{-6} - 4.0×10^{-4} (5.1×10^{-5})[c]	6.8×10^{-6} - 4.0×10^{-4} (4.8×10^{-5})
Risk from ^{106}Ru	2.6×10^{-8} - 3.5×10^{-6} (3.4×10^{-7})	2.5×10^{-8} - 4.0×10^{-6} (3.1×10^{-7})
Risk from ^{90}Sr	1.6×10^{-7} - 1.9×10^{-5} (1.9×10^{-6})	2.0×10^{-7} - 3.4×10^{-5} (2.7×10^{-6})
Risk from ^{60}Co	2.0×10^{-8} - 1.6×10^{-6} (1.9×10^{-7})	2.0×10^{-8} - 1.2×10^{-6} (1.7×10^{-7})
Total Risk	8.2×10^{-6} - 4.1×10^{-4} (5.7×10^{-5})	7.8×10^{-6} - 4.2×10^{-4} (5.4×10^{-5})
Female		
Risk from ^{137}Cs	5.5×10^{-6} - 3.3×10^{-4} (4.3×10^{-5})	5.4×10^{-6} - 3.0×10^{-4} (3.9×10^{-5})
Risk from ^{106}Ru	2.1×10^{-8} - 2.9×10^{-6} (2.8×10^{-7})	1.8×10^{-8} - 3.7×10^{-6} (2.5×10^{-7})
Risk from ^{90}Sr	1.3×10^{-7} - 1.6×10^{-5} (1.6×10^{-6})	1.4×10^{-7} - 3.2×10^{-5} (2.2×10^{-6})
Risk from ^{60}Co	1.6×10^{-8} - 1.3×10^{-6} (1.5×10^{-7})	1.5×10^{-8} - 1.0×10^{-6} (1.4×10^{-7})
Total Risk	6.7×10^{-6} - 3.4×10^{-4} (4.8×10^{-5})	7.1×10^{-6} - 3.2×10^{-4} (4.4×10^{-5})

[a] The fish consumption rate for Category I individuals is defined by a 95% subjective confidence interval of 7.1-33 kg y^{-1} for males (central value, 17 kg y^{-1}) and 5.7-27 kg y^{-1} for females (central value, 14 kg y^{-1}). See Sections 7 and 13 for more details.

[b] Between 8 and 20% of the total fish consumed is in the form of fish patties.

[c] The range indicates the 95% subjective confidence interval, and the value in parentheses indicates the central value (50th percentile).

14.2 Ingestion of Contaminated Animals from the Oak Ridge Reservation

The possibility exists that off-site individuals have been exposed to radionuclides through the consumption of fish, deer, waterfowl, or turtles that were contaminated inside the Oak Ridge Reservation and either were taken illegally on-site or migrated off-site and were taken legally. These animals could have been contaminated at levels much higher than those estimated for fish in the Clinch River. The following sections describe scenarios for each type of wildlife, including the assumptions made about the amounts ingested, the radionuclide concentrations used for calculations, and the total excess lifetime risk of cancer incidence for each situation. Risks are calculated on a per-meal basis (ingestion of 4-16 oz., or 0.12 to 0.45 kg) and, for larger animals, on a per-animal basis.

14.2.1 Contaminated Fish Migrating from White Oak Lake to the Clinch River

This scenario involves consumption of a resident fish from White Oak Lake that has migrated into the Clinch River and is subsequently caught and eaten. While the probability of this event is low, the excess lifetime risk of cancer incidence that could result from this scenario was estimated, based on a single highly contaminated fish. The risk estimate was based on an individual eating one fish meal ranging from 4 to 16 ounces of fish fillet (0.12 to 0.45 kg).

Several studies have measured the concentration of the radionuclides ^{137}Cs, ^{60}Co, and ^{90}Sr in populations of bluegill (*Lepomis microchirus*) and largemouth bass (*Micropterus salmoides*) in White Oak Lake. The radionuclide concentrations in fish reported in these studies are listed in Table 8.4 and Appendix 14-A (Tables 14A.1 and 14A.2). The highest mean concentrations in fish flesh (76,500 Bq kg^{-1} ^{137}Cs and 2500 Bq kg^{-1} ^{90}Sr) were found in a sample of 4 redhorse (Knobf, 1951). The 1965 Oak Ridge National Laboratory Annual Report gave the highest mean ^{60}Co concentration of 230 Bq kg^{-1} in 5 largemouth bass samples (Auerbach et al., 1966). Calculations were made for the 1948 redhorse samples (Table 14.3; 76,500 Bq kg^{-1} ^{137}Cs and 2,500 Bq kg^{-1} ^{90}Sr) and for a representative value for more recent years (highest mean value reported for bluegill or bass in the 1980s, 3000 Bq kg^{-1} ^{137}Cs, based on Tables 14A.1 and 14A.2).

Table 14.3 Summary of maximum reported radionuclide concentrations in the flesh of fish from White Oak Lake.

	Sample Size	Mean Concentration [Bq kg^{-1}]	Date	Reference
^{137}Cs (Redhorse)	5	76,500	1948	Knobf, 1951
^{90}Sr (Redhorse)	5	2500	1948	Knobf, 1951
^{60}Co(Largemouth bass)	4	230	1965	Auerbach et al., 1966

Based on the 1948 measurements, a single meal of fish from White Oak Lake could have resulted in an excess lifetime risk of cancer incidence of 5.1×10^{-6} to 3.5×10^{-5} (Table 14.4; central value, 1.4×10^{-5}). Based on the highest reported mean value of measurements made in the 1980s, a single fish meal from White Oak Lake could have resulted in a total increased risk of 1.9×10^{-7} to 1.3×10^{-6} (Table 14.4; central value, 5.4×10^{-7}). Thus, depending on when a fish from White Oak Lake was caught and consumed, risks as high as 1×10^{-5} or as low as 1×10^{-7} could have occurred from a single meal.

Table 14.4 Examples of estimated excess lifetime risk of cancer incidence from the consumption of a single meal (4-16 oz.) of fish (flesh only) from White Oak Lake.

Case	Contamination level		95% Subjective Confidence Interval		
	(Bq kg^{-1})		lower limit	central value	upper limit
1	redhorse, 1948				
	^{137}Cs	76,500	4.9×10^{-6}	1.4×10^{-5}	3.3×10^{-5}
	^{90}Sr	2,500	1.8×10^{-7}	6.0×10^{-7}	2.2×10^{-6}
		Total	5.1×10^{-6}	1.4×10^{-5}	3.5×10^{-5}
2	bass and bluegill, 1980s				
	^{137}Cs	3,000	1.9×10^{-7}	5.4×10^{-7}	1.3×10^{-6}

14.2.2 Contaminated Deer Wandering off the Oak Ridge Reservation

This scenario addresses the estimation risk for individuals consuming deer from the Oak Ridge Reservation. Deer were present in small numbers prior to 1969, when the first road kill was reported (Story and Kitchings, 1982). However, when the number of car-deer accidents approached 300 per year, the reservation was opened to controlled hunting. Legal deer hunts were established on the reservation in 1985; all deer killed are checked for contamination and confiscated if field measurements of radioactivity exceed 185 Bq kg^{-1} ^{137}Cs in flesh or 740 Bq kg^{-1} ^{90}Sr in bone (Appendix 14-A). Approximately 2.5% of deer killed between 1985 and 1995 were confiscated. The maximum measured concentrations of radionuclides in flesh were 43,000 Bq kg^{-1} of ^{137}Cs (a two and one-half year old doe taken in 1994 from Copper Ridge, in the vicinity of the Health Physics Research Reactor; Scolfield, 1996; Table 14A.3, deer #409) and 880 Bq kg^{-1} of ^{90}Sr (deer #157 in Table 14A.4, taken in 1993; Scolfield, 1996).

The actual measured concentrations of ^{137}Cs and ^{90}Sr in the two deer described above were used in the risk calculations; a third hypothetical case was based on the confiscation limits. It is expected that, even allowing for some inaccuracy in field measurements, more than 90% of the deer killed in legal hunts would be below the confiscation limits and hence would produce lower levels of risk. The maximum deer are used to estimate the possible exposure that could have occurred from consumption of deer prior to the institution of legal hunts.

Risks were calculated on the basis of a single meal (4-16 oz., or 0.12-0.45 kg) and on the basis of a single animal (over several months or a year). The average mass of a deer (buck) was estimated to be 50-60 kg (Saunders, 1996), of which approximately 30-40% is edible [based on estimates obtained from Herron Packing Company (32%), LL Bean Game and Fish Cookbook (28.7%), and Ron Saunders of the Tennessee Wildlife Resources Agency (41.2%).

On a per-meal basis (Table 14.5), the total excess lifetime risk of cancer incidence ranged from 2.2×10^{-8} to 1.6×10^{-7} (95% subjective confidence interval; central value, 6.0×10^{-8}) for a deer at the confiscation limits to 4.2×10^{-6} to 2.6×10^{-5} (central value, 1.1×10^{-5}) for the highest measurement of ^{137}Cs reported in deer (#409 in 1994). For both of these deer, the dominant contributor to the risk was ^{137}Cs. For case 2 (the highest ^{90}Sr concentration measured in deer; #157 in 1993), the estimated total risk ranged from 1.5×10^{-7} to 1.6×10^{-6} (central value, 4.7×10^{-7}), with more than half of the risk contributed by ^{90}Sr (Table 14.5). The estimated total risks from consumption of an entire deer by one person are approximately 70 times higher than the risks for a single meal. Thus someone consuming meat from a legally obtained deer would have an estimated risk below 1×10^{-7} from a single meal and below 1×10^{-5} from consumption of an entire animal. However, someone consuming meat from an animal that had not been cleared at the check station at a legal hunt could have a total excess lifetime risk of cancer incidence as high as 1×10^{-5} for a single meal or 1×10^{-3} for an entire animal. Based on the frequency of confiscated deer at legal hunts, the probability of illegally obtaining a highly contaminated animal is not expected to be high, although it would depend on when and where the animal was obtained.

Table 14.5 Examples of excess lifetime risk of cancer incidence from the consumption of deer from the Oak Ridge Reservation.

Case	Contamination level (Bq kg^{-1} in flesh)		Single meal (4-16 oz.)	Entire deer[a]
1	Deer #409 (1994)			
	^{137}Cs	43,000	4.2×10^{-6} - 2.5×10^{-5} (1.1×10^{-5})[c]	3.8×10^{-4} - 1.4×10^{-3} (7.2×10^{-4})
	^{90}Sr	380	3.6×10^{-8} - 5.0×10^{-7} (1.3×10^{-7})	3.2×10^{-6} - 2.9×10^{-5} (8.6×10^{-6})
	Total		4.2×10^{-6} - 2.6×10^{-5} (1.1×10^{-5})	3.9×10^{-4} - 1.4×10^{-3} (7.3×10^{-4})
2	Deer #157 (1993)			
	^{137}Cs	720	7.0×10^{-8} - 4.2×10^{-7} (1.8×10^{-7})	6.4×10^{-6} - 2.4×10^{-5} (1.2×10^{-5})
	^{90}Sr	880	8.3×10^{-8} - 1.2×10^{-6} (2.9×10^{-7})	7.5×10^{-6} - 6.7×10^{-5} (2.0×10^{-5})
	Total		1.5×10^{-7} - 1.6×10^{-6} (4.7×10^{-7})	1.4×10^{-5} - 9.1×10^{-5} (3.2×10^{-5})

Table 14.5 (continued)

Case	Contamination level (Bq kg^{-1} in flesh)		Single meal (4-16 oz.)	Entire deer[a]
3	Confiscation limits[b]			
	^{137}Cs	185	1.8×10^{-8} - 1.1×10^{-7} (4.6×10^{-8})	1.7×10^{-6} - 6.1×10^{-6} (3.1×10^{-6})
	^{90}Sr	40	3.8×10^{-9} - 5.3×10^{-8} ($1.3. \times 10^{-8}$)	3.4×10^{-7} - 3.1×10^{-6} (9.0×10^{-7})
	Total		2.2×10^{-8} - 1.6×10^{-7} (6.0×10^{-8})	2.0×10^{-6} - 9.1×10^{-6} (4.0×10^{-6})

[a] Estimated mass, 50-60 kg, of which 30-40% is edible.

[b] Deer were confiscated based on field measurements above 185 Bq kg^{-1} of ^{137}Cs or 740 Bq kg^{-1} of ^{90}Sr. The value of 40 Bq kg^{-1} for ^{90}Sr in flesh is based on a bone-flesh ratio for ^{90}Sr of about 20, estimated from reported laboratory measurements for Oak Ridge deer (Appendix 14-A; observed ratios were typically 20-200). From 1985-1995, approximately 2.5% of the deer killed were confiscated.

[c] The range indicates the 95% subjective confidence interval, and the value in parentheses indicates the central value (50th percentile).

14.2.3 Contaminated Waterfowl Flying off the Oak Ridge Reservation

This scenario addresses the estimate of risk for an individual who consumed waterfowl that migrated off the Oak Ridge Reservation. Waterfowl are known to reside on the reservation, either temporarily during migration or longer term for nesting, so the possibility exists that a contaminated bird could be obtained by a hunter in some other location. Ducks banded at White Oak Lake in the winter of 1952-1953 were reportedly found in other parts of Tennessee and in Alabama, Kentucky, Louisiana, Texas, and Ontario (Krumholz et al., 1953). The primary radionuclide detected in the waterfowl collected from ORNL environments has been ^{137}Cs. Low levels of ^{60}Co were detected in some samples (mostly whole body analyses), and ^{90}Sr was detected in bone samples. Available data for several types of waterfowl are provided in Appendix 14-A (Tables 14A.5, 14A.6, and 14A.7).

The highest observed ^{137}Cs concentrations in waterfowl occurred in a pair of Canada geese that nested regularly on Pond 3524 at the Oak Ridge National Laboratory (ORNL) in 1989. This pond is a small radioactive waste pond located in the ORNL compound and contains relatively high concentrations of radioactivity in its bottom sediment. The geese were feeding on newly emerged vegetative shoots in the pond and accumulated relatively high concentrations of ^{137}Cs. The two geese were sacrificed and were found to have accumulated 150,000 and 105,000 Bq kg^{-1} of ^{137}Cs, respectively, in their breast tissue (Loar et al., 1994b). As a result of these high concentrations of radioactivity, actions were taken to prevent the use of the waste ponds as nesting sites.

The risk estimate was based on 2 cases, the highest measured concentration (150,000 Bq kg^{-1} of ^{137}Cs) in geese residing on the waste pond and the highest average concentration measured in geese residing on White Oak Lake (1200 Bq kg^{-1} of ^{137}Cs). Most geese and all mallards and American coots measured in the 1980s were below 1200 Bq kg^{-1} (Appendix 14-A); concentrations could have been higher in prior years. Risks were calculated on the basis of a single meal (4-16 oz., or 0.12-0.45 kg) and on the basis of a single animal. The average mass of a goose was estimated to be 4.5-5.4 kg (Saunders, 1996), of which approximately 30-35% is edible.

On a per-meal basis (Table 14.6), the total excess lifetime risk of cancer incidence ranged from 1.2×10^{-7} to 7.0×10^{-7} (95% subjective confidence interval; central value, 3.0×10^{-7}) for geese on White Oak Lake in the 1980s to 1.5×10^{-5} to 8.8×10^{-5} (central value, 3.7×10^{-5}) for the highest measurement of ^{137}Cs reported in a Canada goose. The estimated total risks from consumption of an entire goose by one person are approximately 6 times higher than the risks for a single meal (Table 14.6). Thus consumption of highly contaminated waterfowl could result in a risk approaching or exceeding 1×10^{-4}, depending on the amount of meat consumed; given the relatively small size of the waste pond where the highly contaminated birds were obtained, the likelihood of a hunter legally obtaining waterfowl from that site is small. At least for the 1980s, consumption of one bird from the reservation will most likely result in an increased risk of cancer below 1×10^{-6}. The actual risk to a real individual would depend on where and when the waterfowl were obtained and how much was consumed.

Table 14.6 Examples of excess lifetime risk of cancer incidence from the consumption of geese from the Oak Ridge Reservation.

Case	Contamination level (Bq kg^{-1} in flesh)		Single meal (4-16 oz.)	Entire goose[a]
1	^{137}Cs	150,000	1.5×10^{-5} - 8.8×10^{-5} (3.7×10^{-5})[b]	1.1×10^{-4} - 4.1×10^{-4} (2.1×10^{-4})
2	^{137}Cs	1,200	1.2 × 10-7 - 7.0 × 10-7 (3.0 × 10-7)	8.8×10^{-7} - 3.3×10^{-6} (1.7×10^{-6})

[a] Estimated mass, 4.5-5.4 kg, of which 30-35% is edible.

[b] The range indicates the 95% subjective confidence interval, and the value in parentheses indicates the central value (50th percentile).

14.2.4 Contaminated Turtles Migrating from White Oak Lake to the Clinch River

This scenario addresses the estimate of risk for an individual consuming turtles that may have migrated off the Oak Ridge Reservation. The most common species found in White Oak Lake include the pond slider (*Trachemys scripta*), and the common snapping turtle (*Chelydra serpentina*), the latter being the primary edible species. Demographic studies on East Fork Poplar Creek (Task 3 Report of the Oak Ridge Dose Reconstruction) indicated that turtles were sometimes used as occasional food sources, although no one interviewed in the demographic studies of the Clinch River (Section 7 of this report) mentioned turtle consumption.

Radionuclide concentrations were compared in two species of turtles collected in White Oak Lake (the common snapping turtle, primarily omnivorous, and the pond slider, primarily vegetarian; Meyers-Schöne et al., 1993). Cesium-137, ^{60}Co, and ^{90}Sr were measured in twelve turtles of each species; the concentrations were significantly higher than those found in reference pond turtles.

Turtle muscle is the principal tissue eaten. Very little ^{60}Co was detected in muscle of any of the sampled turtles; most was found in the kidney, liver, or egg yolks, all of which are generally discarded. Strontium-90 accumulated primarily in the bone and shell of the turtles. The primary radionuclide found in muscle tissue was ^{137}Cs (Meyers-Schöne et al., 1993), and for this reason the concentrations of ^{137}Cs in turtles were used to evaluate risk. Two of the sliders had ^{137}Cs concentrations much greater than those of any other measured turtles: 502,000 and 33,000 Bq kg^{-1} (wet weight). The average ^{137}Cs concentration in the other 10 sliders was 380 Bq kg^{-1}; for all 12 sliders, the average ^{137}Cs concentration in muscle was 45,000 Bq kg^{-1}. The average ^{137}Cs concentration in flesh (muscle) of the 12 snapping turtles was 40 Bq kg^{-1}. The two highly contaminated sliders are thought to have migrated from another contaminated site to White Oak Lake (Meyers-Schöne et al., 1993).

The risk estimates were based on an individual eating one turtle meal consisting of 4-16 oz. (0.12-0.45 kg) of turtle muscle. Calculations were made for the highest reported ^{137}Cs concentration in turtle flesh (502,000 Bq kg^{-1}), the average slider excluding the two thought to have come from another contaminated site (380 Bq kg-1), and the average snapping turtle (40 Bq kg^{-1}). It is possible that White Oak Lake turtles could have had concentrations greater than 40 or 380 Bq kg^{-1} in the 1950s, for example, but it is unlikely that very many turtles exceeding 500,000 Bq kg^{-1} have occurred, and even less likely that someone catching turtles for consumption near the Oak Ridge Reservation would have encountered such a highly contaminated specimen.

The total excess lifetime risk of cancer incidence for consumption of one meal of turtle meat ranged from 3.9×10^{-9} to 2.3×10^{-8} (95% subjective confidence interval; central value, 1.0×10^{-8}) for snapping turtles (Case 3) to 4.9×10^{-5} to 2.9×10^{-4} (central value, 1.3×10^{-4}) for the highest measurement of ^{137}Cs reported in a turtle (Table 14.7). Thus consumption of a highly contaminated turtle could result in a risk approaching or exceeding 2.9×10^{-4}, depending on the amount of meat consumed. The actual risk to a real individual would depend on where and when the turtle was obtained and how much was consumed.

Table 14.7 Examples of estimated excess lifetime risk of cancer incidence from the consumption of a single meal (4-16 oz.) of turtle (flesh only) from the Oak Ridge Reservation.

Case	Contamination level		95% Subjective Confidence Interval		
		(Bq kg^{-1})	lower limit	central value	upper limit
1	^{137}Cs	502,000	4.9×10^{-5}	1.3×10^{-4}	2.9×10^{-4}
2	^{137}Cs	380	3.7×10^{-8}	9.5×10^{-8}	2.2×10^{-7}
3	^{137}Cs	40	3.9×10^{-9}	1.0×10^{-8}	2.3×10^{-8}

14.3 Summary

This section described two types of internal exposure to radionuclides from the X-10 site that could have occurred historically. These include the ingestion of bones as well as flesh of Clinch River fish, in the form of fish patties made by grinding small fish (Section 14.1), and ingestion of contaminated animals (fish, deer, waterfowl, or turtles) that migrated off the Oak Ridge Reservation (Section 14.2).

Ingestion of fish patties as 8-20% of total fish consumption would result in estimated increases of 15-25% in the doses to the bone and red bone marrow vs. consumption of fish flesh alone, due to the increased amount of ^{90}Sr present in the fish patties. The excess lifetime risk of cancer incidence from ^{90}Sr increases about 40% (based on the 50th percentiles) to 100% (based on the 97.5th percentiles) for people consuming fish patties as well as fish flesh, but the total risk from all radionuclides is essentially the same. This situation is due to the fact that the primary doses and risks from fish consumption are a result of ^{137}Cs contamination. While the doses and risks to the bone and bone marrow are increased by the consumption of additional ^{90}Sr, these doses and risks are small in comparison to the total doses and risks from ^{137}Cs and thus do not contribute significantly to the total doses and risks from consumption of fish.

Estimated excess lifetime risks of cancer incidence for consumption of contaminated animals from the Oak Ridge Reservation are summarized on a per-meal basis (4-16 oz. of contaminated flesh) in Figure 14.1. Based on actual measured radionuclide concentrations (primarily ^{137}Cs), the excess lifetime risk of cancer incidence from consumption of a single meal of highly contaminated Oak Ridge Reservation wildlife could have reached or exceeded 1×10^{-4}. However, more likely concentrations would generally result in risks below 1×10^{-6}. The actual risk to any real individual would depend greatly on the kind of animal consumed, when and where it was obtained, and how much was eaten.

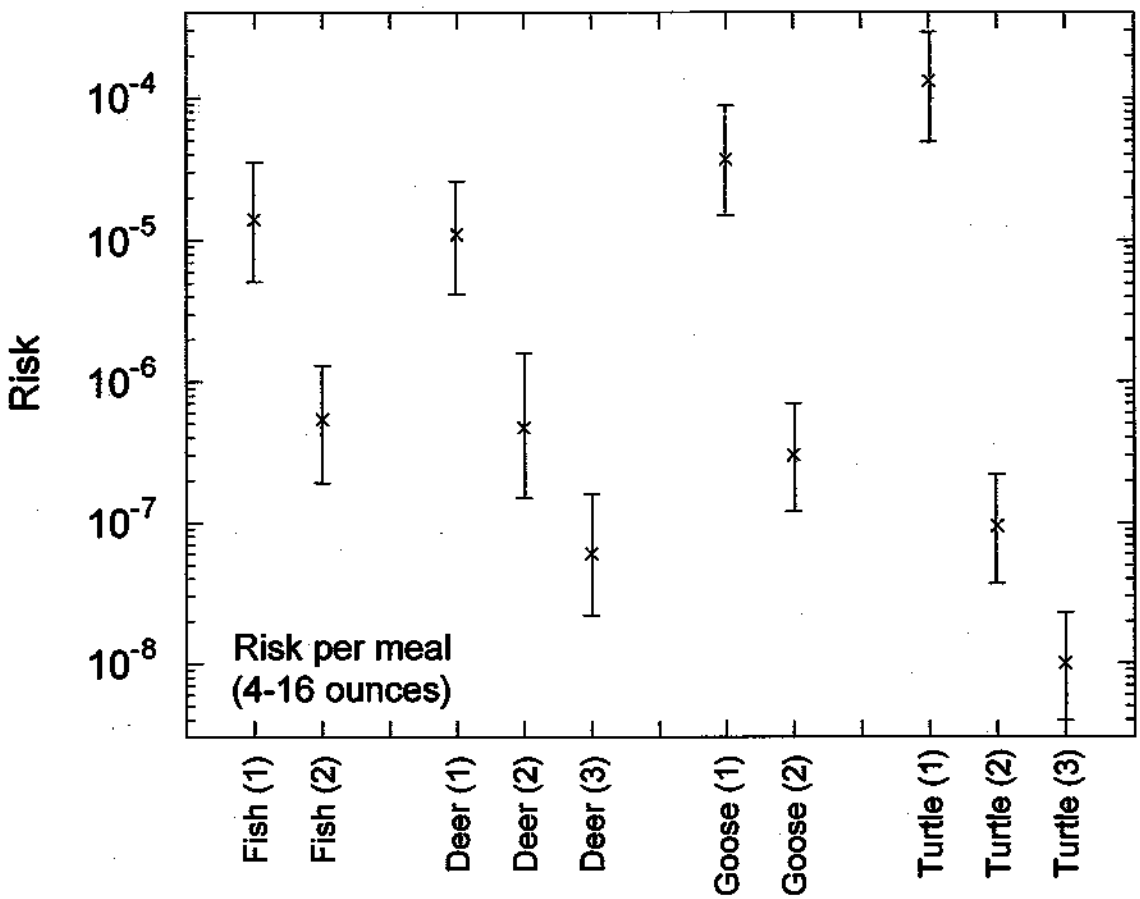

Fig. 14.1 Examples of estimated excess lifetime risk of cancer incidence from the consumption of a single meal (4-16 oz.) of animals from the ORR. Concentrations for each case are given in Tables 14.4-14.7 and in the accompanying text. Cases 1 for fish, geese, and turtles and cases 1 and 2 for deer are based on the highest reported measurements of ^{137}Cs (all animals) and ^{90}Sr (fish and deer) in animal flesh. The remaining cases for fish, geese, and turtles are based on average concentrations measured in the 1980s. Case 3 for deer is based on the confiscation limit for field measurements of radionuclides in legally obtained deer.

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15.0 CONCLUSIONS

Since 1944, radionuclides have been released from the X-10 facility to the Clinch River due to the chemical processing of nuclear materials, the operation of several nuclear reactors, and the use of the X-10 site as a repository for radioactive wastes for the Southeast Region of the United States. Estimates of yearly releases of radioactive materials from the X-10 site were based on a detailed evaluation of historical measurements of radioactive discharges from White Oak Dam and daily flow rates of water. About 5.9×10^{15} Bq (160,000 curies) of various radionuclides were released to the Clinch River over a period of 48 years (1944 to 1991), with the majority of these releases having occurred prior to 1964. Of this material, about 91% was tritium, with the remainder being a mixture of fission and activation products.

The purpose of this investigation was to estimate the radiological doses and health risks to individuals exposed to radioactive materials released to the Clinch River from the X-10 facility. Direct measurement of the amounts of radionuclides taken up by the organs of specific individuals since 1944 is no longer feasible because most of these radionuclides have short residence times in the human body. Therefore, a dose reconstruction has been necessary to determine the magnitude and extent of past exposure and to interpret the health consequences of these exposures. This dose reconstruction relies upon independent evaluation of the amount of radionuclides released, reported environmental measurements, and mathematical models to estimate the magnitude and extent of past exposures, doses, and health risks.

Every effort has been made to ensure a realistic analysis. Uncertainties are explicitly accounted for in all steps of the calculations, including intake or exposure, dose, and risk, and alternative models for dose-response have been included as an uncertain variable. Subjective probability is used in this report, as well as in all other tasks of the Oak Ridge Dose Reconstruction, as the mathematical language for communicating the state of knowledge for each uncertain variable. Uncertainties were propagated through all calculations using Monte Carlo simulation to generate subjective probability distributions composed of numerous alternative realizations of the true values of exposures, doses, and health risks. The extent of uncertainty in exposure, dose, or risk is expressed as a 95% subjective confidence interval and a central estimate. The model components that contribute most to the overall expression of uncertainty were identified through sensitivity analysis. The explicit inclusion in this analysis of uncertainties in dosimetry and dose-response constitutes a level of sophistication beyond that used by the EPA, ICRP, or NCRP.

An initial screening assessment narrowed the scope of the dose reconstruction to the following radionuclides: ^{60}Co, ^{90}Sr, ^{95}Nb, ^{95}Zr, ^{106}Ru, ^{131}I, ^{137}Cs, and ^{144}Ce. Although tritium dominated the total amount of radioactivity released to the Clinch River, the initial screening evaluation indicated that the maximum health risk to off-site residents exposed to tritium for the entire 48-year period would not exceed one-tenth of the ORHASP decision criterion; therefore, tritium was not included among the radionuclides considered for detailed evaluation.

Overall, five exposure pathways were identified as warranting detailed analysis in this task:

- internal irradiation due to the consumption of radionuclides in fish, water, milk, and meat, and

- external irradiation from gamma-emitting radionuclides deposited in shoreline sediment.

In addition, an assessment was performed for the special circumstance of a highly contaminated organism (fish, turtle, waterfowl, or deer) from the X-10 reservation being caught and consumed by someone.

Human exposures were evaluated at four locations: Clinch River Mile (CRM) 20.5 (the confluence of White Oak Creek and the Clinch River), CRM 14 (near the K-25 water intake and the confluence of Grassy Creek), CRM 3.5 (near the Kingston Steam Plant), and CRM 0.0 (the confluence of the Clinch and Tennessee Rivers, near the drinking water intake on the Tennessee River for the City of Kingston). Annual average concentrations of radionuclides in water and sediment were estimated from reported measurements or from estimated discharges across White Oak Dam. A mathematical model (HEC-6-R) was used to simulate the downstream transport and fate of radionuclides released from White Oak Dam and the interaction of these radionuclides with sediment. The HEC-6-R model was calibrated using estimates of the inventory of radionuclides in deposited sediment in various reaches of the Clinch River.

Comparison of modeled water concentrations with monitoring data indicated that a secondary source of contamination to the Clinch River may have occurred after 1963 due to the scouring of contaminated sediment in White Oak Creek Embayment caused by peaking discharges from Melton Hill Dam. Therefore, measured rather than predicted radionuclide concentrations in water were used when available.

Defined reference individuals who may have consumed contaminated fish harvested from the Clinch River were evaluated by gender and by the relative amount of fish consumed. Reference individuals were also described for the other exposure pathways. These individuals were considered to be adults at the time of exposure (for radionuclides other than iodine). For the evaluation of exposures to ^{131}I in milk and drinking water, the assessment targeted the most radiologically sensitive subgroup of the population, children under the age of 15 at the time of exposure.

The highest exposures, doses, and estimated lifetime risks of excess cancer incidence were from the ingestion of contaminated fish. The most highly contaminated fish would have been harvested in the vicinity of CRM 20.5, near Jones Island. At this location the population of fish could include individual fish that migrated out of the mouth of White Oak Creek Embayment or that resided in the unmixed plume of contaminated water coming into the Clinch River from the mouth of White Oak Creek.

Downstream of CRM 20.5, the estimated concentrations of radionuclides in fish decreased rapidly and then remained nearly constant between CRM 14 and CRM 0, where the Clinch River

flows into the Tennessee River at the City of Kingston. The small differences in estimated fish concentrations are consistent with the small differences in water concentrations between these locations. Further downstream, in lower Watts Bar Reservoir, a marked decrease in the concentration of radionuclides in fish occurred due to the mixing and further dilution of contamination from the Clinch River with uncontaminated water and sediment from the Tennessee River.

In the vicinity of Jones Island (CRM 20.5), the highest estimated organ dose was to the bone of an individual who actively consumed fish caught from this location over a period of 48 years (95% subjective confidence interval of 0.067 to 7.4 cSv for males and 0.058 to 7.9 cSv for females). For both males and females, the doses to all other organs were generally within a factor of 2-4 of that estimated for the bone. The similarity among organ doses occurs because the dominant radionuclide is ^{137}Cs, which, once absorbed, is distributed evenly among all tissues of the human body. Doses to bone and red bone marrow are the highest due to the additional contribution from ^{90}Sr.

The excess lifetime risk of cancer incidence associated with exposure was determined primarily by the amount of fish ingested, the decade when fishing occurred, and the location where fishing occurred (specifically, CRM 20.5 vs. other locations). At all locations in the Clinch River, the upper bound on the excess lifetime health risk to a Category I consumer of fish (7.1-33 kg y^{-1} for males, 5.7-27 kg y^{-1} for females) exceeds 1×10^{-4}. At CRM 20.5, the upper bound on the total dose to all organs of a Category I consumer resulted in a lifetime risk of excess cancer exceeding one chance in one thousand (1×10^{-3}). The highest risks occurred as the result of fish consumption between 1944 and 1963. An individual whose exposure to contaminated fish began after 1964 would have received an exposure and dose leading to an excess lifetime risk of cancer below 1×10^{-4}. At locations downstream from CRM 20.5, Category II consumers of fish (2.2-16 kg y^{-1} for males, 1.8-14 kg y^{-1} for females) have risks for which only the upper bound of the 95% confidence interval exceeds 1×10^{-4}. At these downstream locations, the risk estimates for Category III consumers of fish (0.39-4.3 kg y^{-1} for males, 0.32-3.6 kg y^{-1} for females) were clearly below 1×10^{-4}.

The female breast is among the organs receiving the lowest dose, but it has the largest estimated organ risk, due to the high background rate for this type of cancer and the risk model employed. This fact illustrates the importance of using risk as the assessment endpoint in a dose reconstruction, because the organs with the highest doses may not be the organs with the highest risks.

The bioconcentration factor used to estimate the uptake of radionuclides from water to the edible tissues of fish dominates the uncertainty in the risk to Category I consumers of fish. At CRM 20.5, the uncertainty in the risk estimates is about a factor of 10 on either side of the central estimate due to the large uncertainty in the ^{137}Cs concentration in fish at this location. This large uncertainty is influenced both by the bioconcentration factor and by the factor used to account for fish that are migrating out of White Oak Creek Embayment or are resident in the unmixed plume from White Oak Creek. At other

locations, the uncertainty in the ^{137}Cs concentration in fish is due only to the bioconcentration factor, and this reduces the uncertainty in the overall risk estimates to all consumers of fish to a factor of about 6-8 on either side of the central estimate. Uncertainty in the estimated intake of fish was the dominant source of uncertainty for Category III consumers. Uncertainties in the dose-response (risk factors) were important for external exposure and for internal exposure to ^{90}Sr and ^{106}Ru via drinking water, in addition to uncertainties affecting the amount of exposure an individual received.

For Category II consumers of fish (2.2-16 kg y^{-1} for males, 1.8-14 kg y^{-1} for females) at all locations downstream from CRM 20.5, the estimates of doses and risks from the ingestion of fish are comparable to those from external exposure to radionuclides in shoreline sediment and from the consumption of radionuclides in drinking water (at CRM 3.5 and 14). The upper bound on the total risk from all three pathways at these locations is above 1×10^{-4} for all categories of fish consumption. For children exposed to ^{131}I via the consumption of milk or contaminated drinking water, the upper bounds of the 95% subjective confidence intervals are always below 1×10^{-4} (highest value, 2.5×10^{-5}).

For individuals using or residing on Watts Bar Reservoir, the exposures, doses, and risks are substantially lower than they are for individuals using any segment of the Clinch River. Our best estimate is that individual exposures from the past consumption of contaminated fish in Watts Bar Reservoir are 4 to 25 times less than for persons catching fish from the Clinch River.

The radiological doses and excess lifetime cancer risks estimated in this report are incremental increases above those resulting from exposure to natural and other anthropogenic sources of radiation. Nevertheless, for the exposure pathways considered in this task, the doses and risks are not large enough for a commensurate increase in health effects in the population to be detectable, even by the most thorough of epidemiological investigations. In most cases, the estimated organ-specific doses are clearly below the limits of epidemiological detection (1 to 30 cSv) for radiation-induced health outcomes that have been observed following irradiation of large cohorts of individuals exposed either *in utero* (Doll and Wakeford, 1997), as children, or as adults (NRC, 1990; Thompson et al., 1994; Pierce et al., 1996).

Even in the case of Category I consumers of fish (7.1-33 kg y^{-1} for males, 5.7-27 kg y^{-1} for females), the upper confidence limits on the estimated organ-specific doses are below 10 cSv, and the central values are below 1 cSv. The lower confidence limits on these doses are well below levels that have been considered as limits of epidemiological detection in studies of cohorts of other exposed populations. The large uncertainty, combined with the small number of individuals comprising Category I consumers, diminishes the statistical power available to detect a dose response through epidemiological investigation. Therefore, it is unlikely that any observed trends in the incidence of disease in populations that utilized the Clinch River and Lower Watts Bar Reservoir after 1944 could be conclusively attributed to exposure to radionuclides released from the X-10 site, even though this present dose reconstruction study has potentially identified increased individual risks resulting from these exposures.

15.1 References

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The report of project Task 1

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Appendices to the Iodine-131 Report

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• Volume 2A •
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• Volume 6 •
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The report of project Task 7

• Volume 7 •
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