**PERFORMANCE ASSESSMENT FOR
CONTINUING AND FUTURE OPERATIONS
AT SOLID WASTE STORAGE AREA 6**

APPENDICES

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Appendix A

RADIONUCLIDE INVENTORY DATA AND EVALUATION OF UNCERTAINTY IN INVENTORY DATA IN SUPPORT OF THE SWSA 6 PERFORMANCE ASSESSMENT

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Appendix A

RADIONUCLIDE INVENTORY DATA AND EVALUATION OF UNCERTAINTY IN INVENTORY DATA IN SUPPORT OF THE SWSA 6 PERFORMANCE ASSESSMENT

A.1 INTRODUCTION

This appendix provides the radionuclide inventory data used for the Solid Waste Storage Area (SWSA) 6 Performance Assessment (PA). The uncertainties in the radionuclide inventory data are also provided, along with descriptions of the methods used to estimate the uncertainties.

A.2 BACKGROUND

U. S. Department of Energy (DOE) Order 5820.2A, "Management of Low-Level Radioactive Waste," Chapter III, requires field organizations with disposal sites to prepare a site-specific radiological PA for the disposal of low-level radioactive waste (LLW) for the purpose of demonstrating compliance with the performance objectives stated in the order.

ORNL submitted a draft PA in September 1990 to DOE for review. DOE Headquarters' LLW Peer Review Panel completed their preliminary review in March 1991 and issued comments and recommendations to be addressed in the final PA to DOE Headquarters in April 1991. One recommendation of the Peer Review Panel was to determine the potential radiological dose impact due to uncertainty in the SWSA 6 transport and dose models. The uncertainty in the radionuclide inventory was identified as one of seven major model components that influenced simulation results and was evaluated in the 1994 SWSA 6 PA.

The draft PA raised concerns regarding the concentration of several radionuclides disposed at five of the seven SWSA 6 disposal units. The draft radionuclide concentration limits, developed in January 1991, were based upon the results of the draft PA, which covered waste disposed in the period from September 1, 1988, until October 4, 1989. Disposals beyond October 4, 1989, were projected based on past disposal history and operational peaks. Key radionuclides were identified based upon the total activity, volume, and concentration limit. The 1994 SWSA 6 PA covered the period from September 26, 1988, to April 7, 1992. The start date was revised from September 1 to September 26, 1988, to reflect the effective date of DOE Order 5820.2A. Some of the key radionuclides in the draft PA were relatively insignificant in the 1994 SWSA 6 PA. Examples of some of these radionuclides include ^{232}Th in the low-range silos and ^{232}Th , ^{233}U , and ^{237}Np in

Tumulus I and II. This difference was due to (a) errors found in the original inventory data for ^{232}Th in the low-range silos and Tumulus I and II, and (b) lower average concentrations resulting from increases in total volume disposed vs small or no increase in activity. Other key radionuclides identified in the draft PA were still of concern in the 1994 SWSA 6 PA because of the large inventory in disposal. The key radionuclides expected in the 1994 PA were ^{90}Sr , ^{99}Tc , ^{137}Cs , and ^{152}Eu in high range wells; ^{137}Cs , ^{235}U , ^{238}U in fissile wells; and ^{90}Sr in high-range silos, low-range silos, Tumulus I and II, and the Interim Waste Management Facility (IWMF).

In 1997, another PA of SWSA 6 was conducted. This time, the period of the PA covered waste disposed in the period since September 26, 1988. The approach used for the PA was different from that used in 1994. This time, SWSA 6 was modeled in two parts: The first part comprised of modeling the non-IWMF disposal units within SWSA 6, and the second part consisted of modeling the operationally active Interim Waste Management Facility (IWMF). For the non-IWMF disposal units, the radionuclide inventory was taken from the ORNL Waste Tracking System (WTS) database, as described in Sect. A.3. The radionuclide inventory was used to simulate releases from non-IWMF disposal units for comparison with performance objectives. Radionuclides that resulted in a predicted concentration greater than 0.1% of the groundwater limit were considered in the uncertainty analysis. The uncertainty analysis of the inventory for the radionuclides identified by this method is described in Sect. A.4.

A.3 RADIONUCLIDE INVENTORY

The radionuclide inventory data used in this PA were taken from the WTS database. This database is maintained by the ORNL Waste Management and Remedial Action Division (WMRAD). This inventory was grouped by zone (i.e., physical location within SWSA 6) and by type of disposal facility.

To reduce the number of radionuclides required to be evaluated by the PA, the inventory was screened in two steps: First, radionuclides with radiological half-lives less than 5 years were eliminated. Next, screening was performed that compared the estimated radionuclide concentration in the leachate generated in the disposal facility with the concentration limits presented in Table A.1. These limits are based on (1) a dose limit of 4 mrem per year effective dose equivalent and (2) the annual effective dose equivalents from drinking water pathway per unit concentration of radionuclides in water (presented in Table G.7, Appendix G).

The estimated concentration of a radionuclide in leachate in the disposal facility was calculated from the concentration of the radionuclide in solid waste and a dimensionless waste distribution coefficient. The following formula, based on Table C.2 of Appendix C, was used to calculate the liquid (or leachate) concentration:

$$C_l = \frac{C_s}{\epsilon K} \quad , \quad (A.1)$$

where

- C_l = radionuclide concentration in leachate ($\mu\text{Ci/L}$),
- C_s = radionuclide concentration in solid waste ($\mu\text{Ci/L}$),
- ϵ = average open pore void fraction in the waste (dimensionless),
- K = dimensionless distribution coefficient (dimensionless).

For these calculations, it was assumed that $\epsilon = 0.3$.

The radionuclide inventories for each zone and disposal facility are presented in Table A.2 through Table A.13. These tables provide (a) the radioactivity for each nuclide as reported in the WTS database, (b) the nuclide mass, (c) the nuclide inventory per unit facility, and (d) the values used in the nuclide screening [i.e., dimensionless distribution coefficient (K) and solid and liquid concentrations]. The solid concentration is calculated based on the reported radioactivity and the volume of waste disposed (as indicated in the footnotes of the tables). The liquid concentration was calculated by using Eq. (A.1) and was compared with the screening limits presented in Table A.1. Radionuclides that were screened out are indicated with a footnote. The inventory values presented in Table A.2 through Table A.13, with the exception of the radionuclides that were screened out, were used in the calculations for this PA.

A.4 EVALUATION OF UNCERTAINTY IN INVENTORY DATA

This section describes the general considerations involved, and the methods used, to develop estimates of uncertainty in the SWSA 6 inventory data. The SWSA 6 waste inventory data were reviewed twice—the first time in 1993–1994, and the second time in 1996–1997. The first review was very extensive and involved analyzing the waste inventory records and observing how the generators derived their reported data. A flow diagram of the steps taken for estimating the 1994 SWSA 6 PA radionuclide inventories is shown in Fig. A.1. The second review focused mainly on changes in the quality of the reported data since the 1994 SWSA 6 PA. In the second review, the waste inventory records for a number of waste packages were reviewed and several phone calls were made to waste generators in order to verify certain pieces of information.

Table A.1 Concentrations in water for radionuclide screening

Nuclide ^a	Half-life (year)	Concentration ($\mu\text{Ci/L}$) ^b	Nuclide ^a	Half-life (year)	Concentration ($\mu\text{Ci/L}$) ^b
³ H	1.23×10^1	8.7×10^{-2}	²¹⁰ Pb+d	2.23×10^1	7.5×10^{-7}
¹⁰ Be	1.60×10^6	1.2×10^{-3}	²²⁶ Ra+d	1.60×10^3	6.3×10^{-7}
¹⁴ C	5.73×10^3	2.7×10^{-3}	²²⁹ Th+d	7.34×10^3	1.4×10^{-6}
²⁶ Al	7.20×10^5	3.6×10^{-4}	²³⁰ Th	7.54×10^4	1.0×10^{-5}
³⁶ Cl	3.01×10^5	1.8×10^{-3}	²³² Th+d	1.41×10^{10}	1.1×10^{-6}
⁴⁰ K	1.28×10^9	2.9×10^{-4}	²³¹ Pa+d	3.28×10^4	2.1×10^{-7}
⁴¹ Ca	1.03×10^5	4.2×10^{-3}	²³² U+d	6.89×10^1	2.7×10^{-6}
⁶⁰ Co	5.27×10^0	2.0×10^{-4}	²³³ U	1.59×10^5	1.9×10^{-5}
⁵⁹ Ni	7.50×10^4	2.7×10^{-2}	²³⁴ U	2.45×10^5	2.0×10^{-5}
⁶³ Ni	1.00×10^2	9.5×10^{-3}	²³⁵ U	7.04×10^8	2.0×10^{-5}
⁷⁹ Se	6.50×10^4	6.3×10^{-4}	²³⁶ U	2.34×10^7	2.0×10^{-5}
⁸⁷ Rb	4.80×10^{10}	1.1×10^{-3}	²³⁸ U+d	4.47×10^9	2.1×10^{-5}
⁹⁰ Sr+d	2.85×10^1	3.6×10^{-5}	²³⁷ Np	2.14×10^6	1.3×10^{-6}
⁹³ Zr+d	1.53×10^6	2.5×10^{-3}	²³⁸ Pu	8.77×10^1	1.7×10^{-6}
^{93m} Nb	1.36×10^1	1.1×10^{-2}	²³⁹ Pu	2.41×10^4	1.5×10^{-6}
⁹⁴ Nb	2.03×10^4	7.7×10^{-4}	²⁴⁰ Pu	6.56×10^3	1.5×10^{-6}
⁹⁹ Tc	2.13×10^5	3.6×10^{-3}	²⁴¹ Pu	1.44×10^1	8.0×10^{-5}
¹⁰⁷ Pd	6.50×10^6	3.6×10^{-2}	²⁴² Pu	3.76×10^5	1.6×10^{-6}
^{113m} Cd	1.37×10^1	3.3×10^{-5}	²⁴⁴ Pu	8.26×10^7	1.7×10^{-6}
^{121m} Sn	5.50×10^1	3.3×10^{-3}	²⁴¹ Am	4.33×10^2	1.5×10^{-6}
¹²⁶ Sn+d	1.00×10^5	2.7×10^{-4}	^{242m} Am	1.41×10^2	1.5×10^{-6}
¹²⁹ I	1.57×10^7	2.0×10^{-5}	²⁴³ Am	7.37×10^3	1.5×10^{-6}
¹³³ Ba	1.05×10^1	1.6×10^{-3}	²⁴³ Cm	2.85×10^1	2.2×10^{-6}
¹³⁵ Cs	3.00×10^6	7.7×10^{-4}	²⁴⁴ Cm	1.81×10^1	2.7×10^{-6}
¹³⁷ Cs	3.00×10^1	1.1×10^{-4}	²⁴⁵ Cm	8.50×10^3	1.5×10^{-6}
¹⁵¹ Sm	9.00×10^1	1.4×10^{-2}	²⁴⁶ Cm	4.73×10^3	1.5×10^{-6}
¹⁵² Eu	1.33×10^1	8.5×10^{-4}	²⁴⁷ Cm	1.56×10^7	1.6×10^{-6}
¹⁵⁴ Eu	8.80×10^0	5.8×10^{-4}	²⁴⁸ Cm	3.40×10^5	4.0×10^{-7}
¹⁵⁵ Eu	4.96×10^0	3.6×10^{-3}	²⁴⁹ Cf	3.51×10^2	1.2×10^{-6}
¹⁹⁴ Os	6.00×10^0	5.0×10^{-4}	²⁵⁰ Cf	1.31×10^1	2.7×10^{-6}
²⁰⁷ Pb	3.22×10^1	9.8×10^{-4}	²⁵¹ Cf	9.00×10^2	1.1×10^{-6}

^a "d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide.

^b Concentration screening value based on (1) a dose limit of 4 mrem per year effective dose equivalent and (2) the annual effective dose equivalents from drinking water pathway per unit concentration of radionuclides in water (presented in Table G.7, Appendix G).

Table A.2 Radionuclide inventory in Zone 3 high-range silos^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/silo)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H	1.00×10^1	1.04×10^{-3}	2.88×10^{-5}	1.00×10^0	2.33×10^1	7.77×10^1
¹⁴ C	4.11×10^{-2}	9.22×10^{-3}	2.56×10^{-4}	5.50×10^0	9.58×10^{-2}	5.81×10^{-2}
⁴⁰ K	1.74×10^{-3}	2.45×10^2	6.81×10^0	3.50×10^1	4.05×10^{-3}	3.86×10^{-4}
⁶⁰ Co ^c	1.57×10^1	1.39×10^{-2}	3.86×10^{-4}	2.00×10^7	3.66×10^1	6.11×10^{-6}
⁶³ Ni	1.41×10^{-1}	2.29×10^{-3}	6.35×10^{-5}	3.20×10^1	3.29×10^{-1}	3.42×10^{-2}
⁹⁰ Sr	1.35×10^1	9.93×10^{-2}	2.76×10^{-3}	4.40×10^1	3.15×10^1	2.39×10^0
¹⁰⁷ Pd ^c	6.50×10^{-2}	1.26×10^2	3.51×10^0	7.21×10^6	1.52×10^{-1}	7.01×10^{-8}
¹³⁷ Cs	1.62×10^1	1.86×10^{-1}	5.16×10^{-3}	1.00×10^2	3.77×10^1	1.26×10^0
¹⁵² Eu	3.03×10^0	1.75×10^{-2}	4.87×10^{-4}	1.90×10^1	7.07×10^0	1.24×10^0
¹⁵⁴ Eu	1.27×10^0	4.69×10^{-3}	1.30×10^{-4}	1.90×10^1	2.95×10^0	5.18×10^{-1}
¹⁵⁵ Eu	2.90×10^{-2}	6.24×10^{-5}	1.73×10^{-6}	1.90×10^1	6.76×10^{-2}	1.19×10^{-2}
²³⁰ Th ^c	6.50×10^{-7}	3.08×10^{-5}	8.56×10^{-7}	2.70×10^2	1.52×10^{-6}	1.87×10^{-8}
²³² Th	1.36×10^{-4}	1.24×10^3	3.43×10^1	2.70×10^2	3.17×10^{-4}	3.91×10^{-6}
²³³ U ^c	1.00×10^{-7}	1.03×10^{-5}	2.87×10^{-7}	2.80×10^2	2.33×10^{-7}	2.78×10^{-9}
²³⁵ U ^c	9.90×10^{-7}	4.58×10^{-1}	1.27×10^{-2}	2.80×10^2	2.31×10^{-6}	2.75×10^{-8}
²³⁸ U	9.97×10^{-3}	2.97×10^4	8.24×10^2	2.80×10^2	2.32×10^{-2}	2.77×10^{-4}
²³⁹ Pu	1.03×10^{-3}	1.66×10^{-2}	4.60×10^{-4}	2.90×10^2	2.40×10^{-3}	2.76×10^{-5}
²⁴² Pu ^c	2.00×10^{-8}	5.24×10^{-6}	1.46×10^{-7}	2.90×10^2	4.66×10^{-8}	5.36×10^{-10}
²⁴⁴ Cm	5.11×10^{-1}	6.31×10^{-3}	1.75×10^{-4}	2.90×10^2	1.19×10^0	1.37×10^{-2}
Total	6.05×10^1					

^aInventory reported for 36 of 37 high-range silos. Disposals occurred during the period 9/88 to 7/93. Volume reported disposed = 428,940 L.

^bDimensionless distribution coefficients taken from SWSA 6 PA rev. 0 (ORNL 1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

Table A.3 Radionuclide inventory in Zone 5 high-range wells^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/well)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
⁹⁰ Sr	1.33×10^3	9.78×10^0	1.09×10^0	4.40×10^1	2.38×10^5	1.81×10^4
¹³⁷ Cs	1.33×10^3	1.53×10^1	1.70×10^0	1.00×10^2	2.38×10^5	7.95×10^3
Total	2.66×10^3					

^aInventory reported for 9 of 9 high-range wells. Disposals occurred during the period 9/88 to 12/92. Volume reported disposed = 5,580 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

Table A.4 Radionuclide inventory in Zone 5 high-range wells-in-silos^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/well)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
¹⁴ C ^c	1.20×10^{-6}	2.69×10^{-7}	5.49×10^{-9}	5.50×10^0	1.00×10^{-4}	6.09×10^{-5}
⁴¹ Ca	6.80×10^{-1}	7.68×10^0	1.57×10^{-1}	2.32×10^1	5.69×10^1	8.18×10^0
⁶⁰ Co	2.32×10^3	2.05×10^0	4.19×10^{-2}	2.00×10^7	1.94×10^5	3.24×10^{-2}
⁹⁰ Sr	2.08×10^2	1.53×10^0	3.12×10^{-2}	4.40×10^1	1.74×10^4	1.32×10^3
⁹⁹ Tc	4.00×10^{-1}	2.35×10^1	4.80×10^{-1}	6.50×10^0	3.35×10^1	1.72×10^1
¹³⁷ Cs	1.10×10^3	1.27×10^1	2.58×10^{-1}	1.00×10^2	9.21×10^4	3.07×10^3
¹⁵² Eu	1.34×10^3	7.75×10^0	1.58×10^{-1}	1.90×10^1	1.12×10^5	1.97×10^4
¹⁵⁴ Eu	5.79×10^2	2.14×10^0	4.38×10^{-2}	1.90×10^1	4.85×10^4	8.50×10^3
²²⁹ Th	7.50×10^{-3}	3.52×10^{-2}	7.19×10^{-4}	2.70×10^2	6.28×10^{-1}	7.75×10^{-3}
²³² Th	3.73×10^{-5}	3.39×10^2	6.92×10^0	2.70×10^2	3.12×10^{-3}	3.85×10^{-5}
²³⁵ U ^c	1.51×10^{-6}	6.99×10^{-1}	1.43×10^{-2}	2.80×10^2	1.26×10^{-4}	1.50×10^{-6}
²³⁸ U ^c	4.23×10^{-6}	1.26×10^1	2.57×10^{-1}	2.80×10^2	3.54×10^{-4}	4.21×10^{-6}
Total	5.55×10^3					

^aInventory reported for 49 of 53 high-range wells in silos. Disposals occurred during the period 9/88 to 12/92. Volume reported disposed = 11,950 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

Table A.5 Radionuclide inventory in Zone 7 fissile wells^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/well)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
¹³⁷ Cs	4.25×10^1	4.89×10^{-1}	2.44×10^{-1}	1.00×10^2	1.25×10^5	4.17×10^3
²³² Th	1.21×10^{-6}	1.10×10^1	5.50×10^0	2.70×10^2	3.56×10^{-3}	4.40×10^{-5}
²³⁵ U	5.87×10^{-4}	2.72×10^2	1.36×10^2	2.80×10^2	1.73×10^0	2.06×10^{-2}
²³⁸ U	4.89×10^{-3}	1.46×10^4	7.28×10^3	2.80×10^2	1.44×10^1	1.71×10^{-1}
Total	4.25×10^1					

^aInventory reported for 2 of 2 fissile wells. Disposals occurred during the period 3/89 to 12/92. Volume reported disposed = 340 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

Table A.6 Radionuclide inventory in Zone 8 Quadrex trenches^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/trench)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H ^c	8.50×10^{-4}	8.81×10^{-8}	2.20×10^{-8}	1.00×10^0	1.51×10^{-2}	5.05×10^{-2}
⁶⁰ Co ^c	2.60×10^{-1}	2.30×10^{-4}	5.75×10^{-5}	2.00×10^7	4.63×10^0	7.72×10^{-7}
⁹⁰ Sr	4.25×10^{-4}	3.13×10^{-6}	7.81×10^{-7}	4.40×10^1	7.57×10^{-3}	5.74×10^{-4}
¹³⁷ Cs	6.51×10^{-1}	7.48×10^{-3}	1.87×10^{-3}	1.00×10^2	1.16×10^1	3.86×10^{-1}
²³⁰ Th	1.95×10^{-3}	9.24×10^{-2}	2.31×10^{-2}	2.70×10^2	3.47×10^{-2}	4.29×10^{-4}
²³² Th	1.25×10^{-3}	1.14×10^4	2.84×10^3	2.70×10^2	2.23×10^{-2}	2.75×10^{-4}
²³³ U ^c	1.05×10^{-5}	1.08×10^{-3}	2.71×10^{-4}	2.80×10^2	1.87×10^{-4}	2.23×10^{-6}
²³⁵ U	1.70×10^{-4}	7.87×10^1	1.97×10^1	2.80×10^2	3.03×10^{-3}	3.61×10^{-5}
²³⁸ U	1.25×10^{-3}	3.72×10^3	9.30×10^2	2.80×10^2	2.23×10^{-2}	2.65×10^{-4}
²³⁷ Np ^c	4.74×10^{-7}	6.72×10^{-4}	1.68×10^{-4}	2.80×10^2	8.45×10^{-6}	1.01×10^{-7}
²³⁸ Pu	1.70×10^{-3}	9.94×10^{-5}	2.49×10^{-5}	2.90×10^2	3.03×10^{-2}	3.48×10^{-4}
²³⁹ Pu ^c	1.60×10^{-7}	2.57×10^{-6}	6.43×10^{-7}	2.90×10^2	2.85×10^{-6}	3.28×10^{-8}
²⁴⁴ Cm	1.20×10^{-3}	1.48×10^{-5}	3.71×10^{-6}	2.90×10^2	2.14×10^{-2}	2.46×10^{-4}
Total	6.59×10^{-1}					

^aInventory reported for 4 of 5 Quadrex trenches. Disposals occurred during the period 9/88 to 5/89. Volume reported disposed = 56,120 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

Table A.7 Radionuclide inventory in Zone 8 low-range silos^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/silo)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H	1.69×10^1	1.75×10^{-3}	7.30×10^{-5}	1.00×10^0	6.08×10^1	2.03×10^2
¹⁴ C	6.21×10^{-3}	1.39×10^{-3}	5.80×10^{-5}	5.50×10^0	2.23×10^{-2}	1.35×10^{-2}
⁶⁰ Co ^c	5.81×10^0	5.14×10^{-4}	2.14×10^{-5}	2.00×10^7	2.09×10^1	3.48×10^{-6}
⁶³ Ni	1.00×10^{-1}	1.62×10^{-3}	6.75×10^{-5}	3.20×10^1	3.60×10^{-1}	3.75×10^{-2}
⁹⁰ Sr	1.88×10^1	1.38×10^{-1}	5.76×10^{-3}	4.40×10^1	6.76×10^1	5.12×10^0
⁹⁹ Tc	5.74×10^{-2}	3.38×10^0	1.41×10^{-1}	6.50×10^0	2.06×10^{-1}	1.06×10^{-1}
¹³⁷ Cs	1.47×10^1	1.69×10^{-1}	7.04×10^{-3}	1.00×10^2	5.29×10^1	1.76×10^0
¹⁵² Eu	4.67×10^0	2.70×10^{-2}	1.12×10^{-3}	1.90×10^1	1.68×10^1	2.95×10^0
¹⁵⁴ Eu	2.67×10^0	9.89×10^{-3}	4.12×10^{-4}	1.90×10^1	9.60×10^0	1.68×10^0
²²⁶ Ra	6.40×10^{-3}	6.48×10^{-3}	2.70×10^{-4}	1.00×10^2	2.30×10^{-2}	7.67×10^{-4}
²³² Th	6.94×10^{-3}	6.31×10^4	2.63×10^3	2.70×10^2	2.50×10^{-2}	3.08×10^{-4}
²³² U	9.90×10^{-5}	4.63×10^{-6}	1.93×10^{-7}	2.80×10^2	3.56×10^{-4}	4.24×10^{-6}
²³³ U	9.71×10^{-3}	1.00×10^0	4.18×10^{-2}	2.80×10^2	3.49×10^{-2}	4.16×10^{-4}
²³⁵ U	8.31×10^{-4}	3.85×10^2	1.60×10^1	2.80×10^2	2.99×10^{-3}	3.56×10^{-5}
²³⁸ U	1.61×10^{-2}	4.79×10^4	2.00×10^3	2.80×10^2	5.79×10^{-2}	6.89×10^{-4}
²³⁷ Np ^c	4.74×10^{-7}	6.72×10^{-4}	2.80×10^{-5}	2.80×10^2	1.70×10^{-6}	2.03×10^{-8}
²³⁸ Pu	6.82×10^{-4}	3.99×10^{-5}	1.66×10^{-6}	2.90×10^2	2.45×10^{-3}	2.82×10^{-5}
²³⁹ Pu	1.09×10^{-1}	1.75×10^0	7.30×10^{-2}	2.90×10^2	3.92×10^{-1}	4.51×10^{-3}
²⁴² Pu ^c	1.00×10^{-8}	2.62×10^{-6}	1.09×10^{-7}	2.90×10^2	3.60×10^{-8}	4.13×10^{-10}
²⁴¹ Am	9.80×10^{-2}	2.86×10^{-2}	1.19×10^{-3}	2.90×10^2	3.52×10^{-1}	4.05×10^{-3}
²⁴³ Am	2.63×10^{-4}	1.36×10^{-3}	5.68×10^{-5}	2.90×10^2	9.46×10^{-4}	1.09×10^{-5}
²⁴⁴ Cm	3.20×10^{-2}	3.96×10^{-4}	1.65×10^{-5}	2.90×10^2	1.15×10^{-1}	1.32×10^{-3}
²⁴⁹ Cf	1.13×10^{-3}	2.76×10^{-4}	1.15×10^{-5}	3.00×10^2	4.06×10^{-3}	4.51×10^{-5}
UNID ^d	1.51×10^{-2}					
Total	6.40×10^1					

^aInventory reported for 24 of 24 low-range silos. Disposals occurred during the period 9/88 to 4/90. Volume reported disposed = 278,100 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

^dUnidentified. Screened out because reported activity was only 0.02% of total.

Table A.8 Radionuclide inventory in Zone 15 biological trenches^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/trench)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H ^c	1.80×10^{-4}	1.87×10^{-8}	4.66×10^{-9}	1.00×10^0	5.45×10^{-4}	1.82×10^{-3}
¹⁴ C ^c	1.70×10^{-4}	3.81×10^{-5}	9.53×10^{-6}	5.50×10^0	5.15×10^{-4}	3.12×10^{-4}
⁶⁰ Co ^c	2.60×10^{-4}	2.30×10^{-7}	5.75×10^{-8}	2.00×10^7	7.88×10^{-4}	1.31×10^{-10}
⁹⁰ Sr	1.21×10^{-2}	8.90×10^{-5}	2.23×10^{-5}	4.40×10^1	3.67×10^{-2}	2.78×10^{-3}
¹³⁷ Cs	2.86×10^{-3}	3.29×10^{-5}	8.23×10^{-6}	1.00×10^2	8.67×10^{-3}	2.89×10^{-4}
Total	1.52×10^{-2}					

^aInventory reported for 4 of 4 biological trenches. Disposals occurred during the period 3/89 to 3/93. Volume reported disposed = 330,200 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

Table A.9 Radionuclide inventory in Zone 16 asbestos silos^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/silo)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H ^c	1.00×10^{-3}	1.04×10^{-7}	9.42×10^{-9}	1.00×10^0	6.75×10^{-3}	2.25×10^{-2}
¹⁴ C	1.69×10^{-3}	3.80×10^{-4}	3.45×10^{-5}	5.50×10^0	1.14×10^{-2}	6.93×10^{-3}
⁶⁰ Co ^c	8.95×10^{-3}	7.92×10^{-6}	7.20×10^{-7}	2.00×10^7	6.04×10^{-2}	1.01×10^{-8}
⁹⁰ Sr	2.78×10^{-2}	2.04×10^{-4}	1.86×10^{-5}	4.40×10^1	1.88×10^{-1}	1.42×10^{-2}
⁹⁹ Tc	1.37×10^{-3}	8.06×10^{-2}	7.33×10^{-3}	6.50×10^0	9.25×10^{-3}	4.74×10^{-3}
¹³⁷ Cs	8.37×10^{-2}	9.62×10^{-4}	8.75×10^{-5}	1.00×10^2	5.65×10^{-1}	1.88×10^{-2}
¹⁵² Eu	1.00×10^{-3}	5.78×10^{-6}	5.25×10^{-7}	1.90×10^1	6.75×10^{-3}	1.18×10^{-3}
¹⁵⁴ Eu ^c	1.00×10^{-5}	3.70×10^{-8}	3.37×10^{-9}	1.90×10^1	6.75×10^{-5}	1.18×10^{-5}
¹⁹⁴ Os ^c	5.10×10^{-4}	1.64×10^{-6}	1.49×10^{-7}	1.70×10^2	3.44×10^{-3}	6.75×10^{-5}
²³² Th	8.41×10^{-5}	7.65×10^2	6.95×10^1	2.70×10^2	5.68×10^{-4}	7.01×10^{-6}
²³³ U ^c	1.03×10^{-4}	1.06×10^{-2}	9.67×10^{-4}	2.80×10^2	6.95×10^{-4}	8.27×10^{-6}
²³⁵ U ^c	6.94×10^{-7}	3.21×10^{-1}	2.92×10^{-2}	2.80×10^2	4.68×10^{-6}	5.58×10^{-8}
²³⁸ U	4.80×10^{-4}	1.43×10^3	1.30×10^2	2.80×10^2	3.24×10^{-3}	3.85×10^{-5}
²³⁸ Pu ^c	1.00×10^{-6}	5.85×10^{-8}	5.32×10^{-9}	2.90×10^2	6.75×10^{-6}	7.76×10^{-8}
²³⁹ Pu	1.24×10^{-2}	1.99×10^{-1}	1.81×10^{-2}	2.90×10^2	8.37×10^{-2}	9.62×10^{-4}
²⁴¹ Am	1.70×10^{-4}	4.96×10^{-5}	4.51×10^{-6}	2.90×10^2	1.15×10^{-3}	1.32×10^{-5}
²⁴⁴ Cm ^c	3.22×10^{-5}	3.98×10^{-7}	3.62×10^{-8}	2.90×10^2	2.17×10^{-4}	2.50×10^{-6}
Total	1.38×10^{-1}					

^aInventory reported for 11 of 11 asbestos silos. Disposals occurred during the period 2/89 to 9/91. Volume reported disposed = 148,200 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

Table A.10 Radionuclide inventory in Zone 20 biological trenches^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/trench)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
⁶⁰ Co ^c	3.52×10^{-7}	3.12×10^{-10}	1.56×10^{-10}	2.00×10^7	3.13×10^{-6}	5.22×10^{-13}
⁹⁰ Sr	5.88×10^{-3}	4.32×10^{-5}	2.16×10^{-5}	4.40×10^1	5.23×10^{-2}	3.96×10^{-3}
¹³⁷ Cs ^c	3.41×10^{-4}	3.92×10^{-6}	1.96×10^{-6}	1.00×10^2	3.03×10^{-3}	1.01×10^{-4}
Total	6.22×10^{-3}					

^aInventory reported for 2 of 2 biological trenches. Disposals occurred during the period 9/88 to 5/89. Volume reported disposed = 112,400 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

Table A.11 Radionuclide inventory in Zone 26 Tumulus II^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/vault)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H	1.51×10^0	1.56×10^{-4}	7.15×10^{-7}	1.00×10^0	2.54×10^0	8.46×10^0
¹⁴ C	4.24×10^{-3}	9.51×10^{-4}	4.34×10^{-6}	5.50×10^0	7.12×10^{-3}	4.32×10^{-3}
⁶⁰ Co ^c	3.60×10^0	3.19×10^{-4}	1.45×10^{-6}	2.00×10^7	6.05×10^0	1.01×10^{-6}
⁹⁰ Sr	2.81×10^0	2.07×10^{-2}	9.43×10^{-5}	4.40×10^1	4.72×10^0	3.58×10^{-1}
⁹⁹ Tc ^c	5.32×10^{-4}	3.13×10^{-2}	1.43×10^{-4}	6.50×10^0	8.94×10^{-4}	4.58×10^{-4}
¹³⁷ Cs	7.66×10^0	8.80×10^{-2}	4.02×10^{-4}	1.00×10^2	1.29×10^1	4.29×10^{-1}
¹⁵² Eu	1.08×10^{-1}	6.24×10^{-4}	2.85×10^{-6}	1.90×10^1	1.81×10^{-1}	3.18×10^{-2}
¹⁵⁴ Eu	1.08×10^{-1}	4.00×10^{-4}	1.83×10^{-6}	1.90×10^1	1.81×10^{-1}	3.18×10^{-2}
¹⁹⁴ Os	9.52×10^{-2}	3.06×10^{-4}	1.40×10^{-6}	1.70×10^2	1.60×10^{-1}	3.14×10^{-3}
²³² Th	5.02×10^{-4}	4.56×10^3	2.08×10^1	2.70×10^2	8.43×10^{-4}	1.04×10^{-5}
²³² U ^c	1.10×10^{-6}	5.14×10^{-8}	2.35×10^{-10}	2.80×10^2	1.85×10^{-6}	2.20×10^{-8}
²³³ U	4.18×10^{-2}	4.32×10^0	1.97×10^{-2}	2.80×10^2	7.02×10^{-2}	8.36×10^{-4}
²³⁵ U ^c	6.35×10^{-6}	2.94×10^0	1.34×10^{-2}	2.80×10^2	1.07×10^{-5}	1.27×10^{-7}
²³⁸ U	3.49×10^{-3}	1.04×10^4	4.74×10^1	2.80×10^2	5.86×10^{-3}	6.98×10^{-5}
²³⁷ Np ^c	2.37×10^{-7}	3.36×10^{-4}	1.54×10^{-6}	2.80×10^2	3.98×10^{-7}	4.74×10^{-9}
²³⁸ Pu	5.77×10^{-4}	3.37×10^{-5}	1.54×10^{-7}	2.90×10^2	9.69×10^{-4}	1.11×10^{-5}
²³⁹ Pu	8.55×10^{-3}	1.37×10^{-1}	6.28×10^{-4}	2.90×10^2	1.44×10^{-2}	1.65×10^{-4}
²⁴¹ Am	1.25×10^{-2}	3.64×10^{-3}	1.66×10^{-5}	2.90×10^2	2.10×10^{-2}	2.41×10^{-4}
²⁴⁴ Cm	1.94×10^{-2}	2.40×10^{-4}	1.09×10^{-6}	2.90×10^2	3.26×10^{-2}	3.75×10^{-4}
Total	1.60×10^1					

^aInventory reported for 219 of 220 vaults. Disposals occurred during the period 10/90 to 3/92. Volume reported disposed = 595,190 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

Table A.12 Radionuclide inventory in Zone 27 Tumulus I^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/vault)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H	2.70×10^0	2.80×10^{-4}	9.61×10^{-7}	1.00×10^0	3.63×10^0	1.21×10^1
¹⁴ C	3.83×10^{-2}	8.59×10^{-3}	2.95×10^{-5}	5.50×10^0	5.15×10^{-2}	3.12×10^{-2}
⁶⁰ Co ^c	1.04×10^1	9.20×10^{-4}	3.16×10^{-6}	2.00×10^7	1.40×10^1	2.33×10^{-6}
⁶³ Ni	2.02×10^{-1}	3.27×10^{-3}	1.13×10^{-5}	3.20×10^1	2.72×10^{-1}	2.83×10^{-2}
⁹⁰ Sr	1.20×10^1	8.82×10^{-2}	3.03×10^{-4}	4.40×10^1	1.61×10^1	1.22×10^0
⁹⁹ Tc	7.31×10^{-2}	4.30×10^0	1.48×10^{-2}	6.50×10^0	9.83×10^{-2}	5.04×10^{-2}
¹³⁷ Cs	2.47×10^1	2.84×10^{-1}	9.76×10^{-4}	1.00×10^2	3.32×10^1	1.11×10^0
¹⁵² Eu	8.56×10^{-1}	4.95×10^{-3}	1.70×10^{-5}	1.90×10^1	1.15×10^0	2.02×10^{-1}
¹⁵⁴ Eu	1.13×10^0	4.19×10^{-3}	1.44×10^{-5}	1.90×10^1	1.52×10^0	2.67×10^{-1}
²¹⁰ Pb	1.71×10^{-2}	2.24×10^{-4}	7.68×10^{-7}	2.29×10^2	2.30×10^{-2}	3.35×10^{-4}
²²⁶ Ra	1.70×10^{-3}	1.72×10^{-3}	5.91×10^{-6}	1.00×10^2	2.29×10^{-3}	7.62×10^{-5}
²³² Th	6.21×10^{-4}	5.65×10^3	1.94×10^1	2.70×10^2	8.35×10^{-4}	1.03×10^{-5}
²³² U ^c	1.16×10^{-4}	5.42×10^{-6}	1.86×10^{-8}	2.80×10^2	1.56×10^{-4}	1.86×10^{-6}
²³³ U	4.87×10^{-2}	5.03×10^0	1.73×10^{-2}	2.80×10^2	6.55×10^{-2}	7.80×10^{-4}
²³⁴ U ^c	1.00×10^{-8}	1.60×10^{-6}	5.51×10^{-9}	2.80×10^2	1.35×10^{-8}	1.60×10^{-10}
²³⁵ U ^c	1.51×10^{-4}	6.99×10^1	2.40×10^{-1}	2.80×10^2	2.03×10^{-4}	2.42×10^{-6}
²³⁸ U	7.63×10^{-3}	2.27×10^4	7.80×10^1	2.80×10^2	1.03×10^{-2}	1.22×10^{-4}
²³⁷ Np	8.00×10^{-3}	1.13×10^1	3.90×10^{-2}	2.80×10^2	1.08×10^{-2}	1.28×10^{-4}
²³⁸ Pu	5.10×10^{-4}	2.98×10^{-5}	1.02×10^{-7}	2.90×10^2	6.86×10^{-4}	7.89×10^{-6}
²³⁹ Pu	5.10×10^{-2}	8.20×10^{-1}	2.82×10^{-3}	2.90×10^2	6.86×10^{-2}	7.89×10^{-4}
²⁴⁰ Pu	6.80×10^{-3}	2.98×10^{-2}	1.02×10^{-4}	2.90×10^2	9.15×10^{-3}	1.05×10^{-4}
²⁴² Pu ^c	2.30×10^{-7}	6.02×10^{-5}	2.07×10^{-7}	2.90×10^2	3.09×10^{-7}	3.56×10^{-9}
²⁴¹ Am	6.06×10^{-2}	1.77×10^{-2}	6.07×10^{-5}	2.90×10^2	8.15×10^{-2}	9.37×10^{-4}
²⁴³ Am	1.22×10^{-2}	6.13×10^{-2}	2.11×10^{-4}	2.90×10^2	1.64×10^{-2}	1.89×10^{-4}
²⁴³ Cm ^c	1.11×10^{-5}	2.15×10^{-7}	7.39×10^{-10}	2.90×10^2	1.49×10^{-5}	1.72×10^{-7}
²⁴⁴ Cm	2.62×10^{-2}	3.24×10^{-4}	1.11×10^{-6}	2.90×10^2	3.52×10^{-2}	4.05×10^{-4}
²⁴⁹ Cf	1.01×10^{-2}	2.46×10^{-3}	8.47×10^{-6}	3.00×10^2	1.36×10^{-2}	1.51×10^{-4}
MFP ^d	3.30×10^{-4}					
UNID ^e	1.02×10^{-2}					
Total	5.24×10^1					

^aInventory reported for 291 of 291 vaults. Disposals occurred between the period 9/88 to 12/90. Volume reported disposed = 743,290 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

^dMixed fission products. Screened out because reported activity was only 0.006% of total.

^eUnidentified. Screened out because reported activity was only 0.02% of total.

Table A.13 Radionuclide inventory in Zone 30 low-range silos^a

Nuclide	Radioactivity (Ci)	Mass (g)	Inventory (g/silo)	K ^b	Concentration (μCi/L)	
					Solid	Liquid
³ H	9.65×10^0	1.00×10^{-3}	9.52×10^{-6}	1.00×10^0	8.81×10^0	2.94×10^1
¹⁴ C	6.92×10^{-2}	1.55×10^{-2}	1.48×10^{-4}	5.50×10^0	6.32×10^{-2}	3.83×10^{-2}
³⁶ Cl ^c	9.00×10^{-6}	2.73×10^{-4}	2.60×10^{-6}	1.00×10^0	8.22×10^{-6}	2.74×10^{-5}
⁴⁰ K ^c	2.77×10^{-4}	3.91×10^1	3.73×10^{-1}	3.50×10^1	2.53×10^{-4}	2.41×10^{-5}
⁶⁰ Co ^c	6.61×10^0	5.85×10^{-3}	5.57×10^{-5}	2.00×10^7	6.04×10^0	1.01×10^{-6}
⁹⁰ Sr	9.26×10^0	6.81×10^{-2}	6.49×10^{-4}	4.40×10^1	8.46×10^0	6.41×10^{-1}
⁹⁹ Tc ^c	2.70×10^{-4}	1.59×10^{-2}	1.51×10^{-4}	6.50×10^0	2.47×10^{-4}	1.26×10^{-4}
¹³⁷ Cs	1.25×10^1	1.43×10^{-1}	1.36×10^{-3}	1.00×10^2	1.14×10^1	3.79×10^{-1}
¹³³ Ba ^c	1.19×10^{-5}	4.76×10^{-8}	4.53×10^{-10}	6.50×10^1	1.09×10^{-5}	5.57×10^{-7}
¹⁵² Eu	5.00×10^0	2.89×10^{-2}	2.75×10^{-4}	1.90×10^1	4.57×10^0	8.01×10^{-1}
¹⁵⁴ Eu	4.57×10^0	1.69×10^{-2}	1.61×10^{-4}	1.90×10^1	4.17×10^0	7.32×10^{-1}
¹⁵⁵ Eu	2.81×10^{-1}	6.04×10^{-4}	5.76×10^{-6}	1.90×10^1	2.57×10^{-1}	4.50×10^{-2}
²²⁶ Ra	5.50×10^{-4}	5.57×10^{-4}	5.30×10^{-6}	1.00×10^2	5.02×10^{-4}	1.67×10^{-5}
²²⁹ Th	1.70×10^{-4}	7.98×10^{-4}	7.60×10^{-6}	2.70×10^2	1.55×10^{-4}	1.92×10^{-6}
²³⁰ Th	1.17×10^{-3}	5.55×10^{-2}	5.28×10^{-4}	2.70×10^2	1.07×10^{-3}	1.32×10^{-5}
²³² Th	3.79×10^{-3}	3.44×10^4	3.28×10^2	2.70×10^2	3.46×10^{-3}	4.27×10^{-5}
²³² U ^c	1.64×10^{-4}	7.66×10^{-6}	7.30×10^{-8}	2.80×10^2	1.50×10^{-4}	1.78×10^{-6}
²³³ U	6.15×10^{-2}	6.35×10^0	6.05×10^{-2}	2.80×10^2	5.62×10^{-2}	6.69×10^{-4}
²³⁴ U ^c	1.21×10^{-4}	1.94×10^{-2}	1.84×10^{-4}	2.80×10^2	1.11×10^{-4}	1.32×10^{-6}
²³⁵ U ^c	5.57×10^{-5}	2.58×10^1	2.46×10^{-1}	2.80×10^2	5.09×10^{-5}	6.06×10^{-7}
²³⁶ U ^c	6.85×10^{-5}	1.06×10^0	1.01×10^{-2}	2.80×10^2	6.26×10^{-5}	7.45×10^{-7}
²³⁸ U	2.88×10^{-2}	8.57×10^4	8.16×10^2	2.80×10^2	2.63×10^{-2}	3.13×10^{-4}
²³⁷ Np	1.17×10^{-3}	1.66×10^0	1.58×10^{-2}	2.80×10^2	1.07×10^{-3}	1.27×10^{-5}
²³⁸ Pu	8.61×10^{-4}	5.04×10^{-5}	4.80×10^{-7}	2.90×10^2	7.86×10^{-4}	9.04×10^{-6}
²³⁹ Pu	2.40×10^{-2}	3.86×10^{-1}	3.67×10^{-3}	2.90×10^2	2.19×10^{-2}	2.52×10^{-4}
²⁴⁰ Pu ^c	7.22×10^{-7}	3.17×10^{-6}	3.02×10^{-8}	2.90×10^2	6.59×10^{-7}	7.58×10^{-9}
²⁴² Pu	1.00×10^{-3}	2.62×10^{-1}	2.49×10^{-3}	2.90×10^2	9.13×10^{-4}	1.05×10^{-5}
²⁴¹ Am	2.37×10^{-2}	6.91×10^{-3}	6.58×10^{-5}	2.90×10^2	2.16×10^{-2}	2.49×10^{-4}
²⁴³ Am	1.60×10^{-2}	8.05×10^{-2}	7.66×10^{-4}	2.90×10^2	1.46×10^{-2}	1.68×10^{-4}
²⁴⁴ Cm	1.15×10^{-2}	1.42×10^{-4}	1.35×10^{-6}	2.90×10^2	1.05×10^{-2}	1.21×10^{-4}
²⁴⁹ Cf	1.01×10^{-3}	2.47×10^{-4}	2.35×10^{-6}	3.00×10^2	9.23×10^{-4}	1.03×10^{-5}
Total	4.81×10^1					

^aInventory reported for 105 of 105 low-range silos. Disposals occurred during the period 9/88 to 12/93. Volume reported disposed = 1,094,900 L.

^bDimensionless distribution coefficients taken from ORNL (1994).

^cRadionuclide screened out since estimated liquid concentration is less than the screening value.

For the period of 1989 to 1994, the data recorded on form UCN16114, "Log-In Data Sheet for Generators of LLW" and form UCN2822, "Request for Disposal or Storage of Radioactive Solid Waste or Special Materials" were reviewed. (The form currently used at ORNL in lieu of UCN16114 and UCN2822 is UCN2109, "Waste Item Description.") The review revealed several pieces of information that relate to radionuclide activity. This information, its validity, and its usefulness in terms of activity estimation and reporting by the waste generators, is discussed in Sects. A.4.1 through A.4.6. Sections A.4.7 and A.4.8 describe the method used for the 1994 PA to determine the most probable and lower/upper bound activities for the waste inventory. This method was used to provide a more robust estimate of radionuclide activities for cases where a viable exposure rate measurement of the waste package was available. For cases where the measured exposure rates were not viable (e.g., for alpha and beta decay radionuclides), bounding and expert opinion were used to estimate the activities.

In this PA, a change in the PA modeling approach resulted in the need for updating the inventory uncertainty estimates for the SWSA 6 disposal facilities, excluding the IWMF. Thus, the non-IWMF

Fig. A.1 Flow diagram of the uncertainty estimation method used for the 1994 SWSA 6 PA.

inventory uncertainties were reviewed and updated to account for changes since the last PA. The changes and uncertainty updating method are described in Sect. A.5.

A.4.1 Principal Radionuclides

For most facilities, facility operations personnel inferred the listed radionuclides based upon their knowledge of the facility operation or the material being processed. In general, it cannot be said with certainty that any specific package contained the specific radionuclide(s) reported. It also cannot be stated that a package did not contain additional radionuclides [other than the one(s) reported]. On rare occasions, the waste was sampled and appropriate analyses were performed to specifically identify the radionuclides present. The identification of the principal nuclide was straightforward for certain facilities that handled only specific nuclides and where the waste represented process waste. However, for many facilities, the waste form consisted of contaminated components or irradiated materials that likely contained several different radionuclides. In these cases, the principal nuclide identified as being present in each waste package was the nuclide or nuclides that were considered by the generator to be the most significant. This significance was typically based upon a nuclide's activity and its radiological half-life. An example of this is the irradiated metal waste (stainless-steel-clad hulls) from Building 3525. A variety of activation products were present in the metal shortly after irradiation, but ^{60}Co was reported as the only principal nuclide because it is most significant in terms of activity, half-life, and radiation energy. In the case of Building 3517, the principal nuclides were listed as ^{137}Cs and ^{90}Sr . Operations personnel based these choices on the general operating history of the facility. Even though source production processes were operating in different hot cells, all packages of waste from the facility were considered to be composed of equal activities of ^{137}Cs and ^{90}Sr .

A.4.2 Radiation Survey Data for Waste Packages

Radiation survey data were obtained on waste packages using either in-cell probes or portable ion chambers. For the highly radioactive packages, in-cell probes were typically used whenever available. These measurements were made by facility operators and were usually taken in close proximity to the waste package. The distance was usually within 1 ft from the surface of the package, but, in the case of the "5-gal lard can" typically used by most isotopes facilities for highly radioactive in-cell waste, this distance would have been within 1 ft from the center of the can. Note that the in-cell probes were calibrated only prior to installation and most have been in place for many years. The portable survey meters are calibrated semiannually by Radiation Standards and Calibration Laboratory personnel to an accuracy of 10%.

Some facilities did not have in-cell probes and were forced to rely on methods by which portable ion chambers were used in the measurement of large activity packages. In these cases, each facility used a method unique to the facility to obtain the measurements. Because of limitations imposed by the facility design and the range of the available portable instruments, measurements were taken at

distances typically ranging from 3 to 10 ft. The distance to the container was also somewhat uncertain. In these cases, however, facility personnel corrected the dose rate to a 1-ft reading using an inverse-square relationship (based upon the assumed distance from the dose point to the surface of the package) prior to multiplying by the conversion factor to determine the curie content.

For the low-activity waste packages, most measurements were made by Radiation Protection personnel using portable ion chambers. In the majority of these cases, dose rates were taken on small plastic bags of contaminated materials in very close proximity to the package surface, if not on contact. Particularly in these cases, the measurement was taken at several locations along the periphery of the bag. The value recorded was the highest dose rate observed, not the average dose rate.

A.4.3 Quantity of the Principal Nuclide

Most ORNL waste generators estimated the quantity of radioactive material in waste packages by multiplying the beta-gamma dose rate (as discussed above) for each waste packet by the conversion factors listed in the instructions for completing form UCN-16114, "Log-In Data Sheet for Generators of LLW." If the waste was considered to be composed of a single radionuclide, the activity so calculated would be assigned to the principal radionuclide. If the generator considered that multiple radionuclides were present (as discussed above), the activity was divided equally among the radionuclides. Facilities such as Building 3517 split the total activity equally between ^{137}Cs and ^{90}Sr . As a general rule, facilities who reported both ^{137}Cs and ^{90}Sr assigned half of the total activity to each.

The 1-MeV photon dose-rate-to-curie conversion factor was typically used in assigning activity to packages that are thought to contain radionuclides which emit only alphas, betas, or very low-energy gammas or X rays, such as ^3H , ^{14}C , ^{63}Ni , ^{90}Sr , ^{99}Tc , ^{233}U , ^{238}U , ^{239}Pu , ^{241}Am , and ^{243}Am . In certain cases however, typically those involving disposal of fissile material, the activity values are based upon those found on the UCN2681, "ORNL Nuclear Materials Intralaboratory Transfer."

A.4.4 Radiation Survey Data for Waste Carriers

Initially, radiation survey data for waste carriers were considered tools to verify the estimated package activities. The manner in which surveys were taken and recorded varied considerably from package to package and facility to facility. Radiation protection personnel typically located and recorded the "hot spot" dose rate on the carrier, even though the survey was not always identified as such. The hot spot was usually a streaming pathway of some sort, associated with a joint or seam in the cask. For these reasons, calculating source activity in a package from the carrier dose rates was judged to be of doubtful value and was not used in the uncertainty analysis.

A.4.5 Package Weight

The weight of each package is also required to be entered on the UCN16114, and the total weight is required for the UCN2822. Facilities typically do not weigh each package of waste. The package weight is estimated by facility personnel based upon their knowledge of the package contents.

A.4.6 Shortcomings of the ORNL Exposure-Rate-to-Curie Conversion Factors

The table of conversion factors listed on the back of the UCN16114 is provided to facilitate determination of the package activity based upon the measured exposure rate. This exposure rate to curie conversion factor was calculated at a distance of 1 ft for a 1-Ci point source emitting a single 1 MeV photon per disintegration. This is the familiar $6 C E n$ formula, where $E = 1$ MeV per photon and $n = 1$ photon per disintegration. Thus, the exposure rate at 1 ft from a 1-Ci point source is approximately 6 R/h. Conversely, an exposure rate of approximately 1 R/h at 1 ft is attributed to a 0.17 Ci point source. Thus, to determine the activity (in curies) of a point source, a conversion factor of 0.17 Ci per R/h (as used on the UCN16114) is multiplied by the exposure rate, in R/h, measured at 1 ft.

Many potential sources of error are possible when using this conversion factor. Some typical errors might be:

- The radionuclide emits a gamma ray with an energy other than 1 MeV (E).
- The radionuclide emits a gamma ray with an abundance other than 1.0 (n).
- The exposure rate measurement was not taken at 1 ft.
- The waste package was not a point source.
- The radionuclide only emits alpha or beta particles, or soft X rays.
- The waste package provides self-shielding.

The first two factors are evident when considering the gamma factors for common radionuclides as listed in The Health Physics and Radiological Health Handbook (Shleien 1989). Table A.14 shows the exposure rates at 1 ft for a 1 Ci point source of each of the specified radionuclides.

**Table A.14 Exposure rates at 1 ft for
a 1-Ci point source**

Radionuclide	Exposure rate (R/h)
¹³⁷ Cs	3.55
⁶⁰ Co	14.21
¹⁵² Eu	6.24
¹⁵⁴ Eu	6.67
¹⁹² Ir	5.17

The ORNL conversion factor is based upon a exposure rate of 6 R/h. The above table indicates, for example, that a 1-Ci ^{60}Co point source gives a exposure rate which is a factor of 2.4 times greater than would be estimated from the ORNL factor. On the other hand, 3.55 R/h would be observed for a 1-Ci source of ^{137}Cs . This value is only 59% of the 6 R/h used to calculate the ORNL conversion factor. The inverse of the factors in the table above would be used to determine the curie content of the specific radionuclide associated with a measured exposure rate (at 1 ft). Using the ORNL conversion factor (0.17 Ci per R/h), the ^{60}Co activity would be overestimated by a factor of 2.4; the ^{137}Cs activity would be underestimated by a factor of 1.7. Activities of ^{152}Eu , ^{154}Eu , and ^{192}Ir would be more accurately represented (within 15%) using the ORNL conversion factor.

The above factors are, as indicated, for a distance of 1 ft from a point source. For a point source, measurements taken at other distances, if not corrected to a 1-ft reading, could introduce significant errors in the activity estimates, particularly for surveys taken in close proximity to the source. Measurements taken at distances less than 1 ft from the source would lead to overestimates of source activity, while a measurement at greater than 1 ft would give an underestimate. These errors are in addition to those mentioned above for specific radionuclides.

In the field however, true point sources are rarely encountered. High activity radioactive materials are often disposed of in 5-gal lard cans. The cans are typically wrapped in several layers of plastic to avoid contamination of the carrier. The physical dimensions of the can and plastic wrap allow a measurement no closer than a few inches from the wall of the can. The exposure rate from a non-point source, such as the lard can, will begin to approach that from a point source as the distance from the source increases. The use of a point-source response dictates that when this approximation is made, the distance used must be relative to the center of the can, not the distance to the wall of the can. In the case of the lard can, measurements made with incell probes are typically made with the can near the detector, perhaps several inches from the surface of the can. With the can radius of 5 in., the probe could be placed perhaps 3 to 5 in. from the wall of the can, which would be 8 to 10 in. from the center of the can. This distance would result in a higher exposure rate than would be measured at 1 ft, which, for a point source, would overestimate the activity by factors of 1.4 to 2.3.

In those cases where the exposure-rate-to-curie conversion factor was applied to a survey measurement taken at a distance of 1 ft from the surface of the can instead of the center of the can, the resulting activity would be underestimated. In the case of a measurement at 1 ft from the surface of a lard can, since the distance to the center of the can would be 17 in., the resulting activity would be underestimated by a factor of 2.0. For larger waste containers, such as the 55-gal drum, this underestimation would be larger depending on the particulars of the specific situation. For a 55-gal drum measured at 1 ft from the drum surface, the curie content would be underestimated by a factor of 3.8.

At small distances, waste containers do not behave like a point source. A small distance is dependent upon the size of the particular container: For the lard can, this distance would be approximately 1.5 ft measured from the center of the can. Surveys taken closer than this distance would result in higher exposure rates per unit activity than would be observed for a point source at the same distance. At 8 in. from the center of the can (3 in. from the wall), the effects due to

geometry give a exposure rate higher by 1.2 than that for a point source. In this case, the geometry effect would result in an underestimation of the curie content by this factor.

Another impact of the non-point source geometry is that of self absorption of the source and attenuation by the walls of the container. These effects cause an underestimation of the package activity, the significance of which is dependent upon the density and material composition of the waste, the size of the package, and the energy of the photon. Most waste packages have a fairly low density, typically from 0.2 to 1.0 g/cm³. The impact is greater with increasing density and size and decreasing photon energy. For ¹³⁷Cs in a lard can, for example, the underestimation varies from a factor of 1.1 to 1.3 as the density increases from 0.2 to 1.0 g/cm³. For the larger 55-gal drum, the activity would be underestimated by a factor of 1.3 to 2.1. For lower energy photon emitters (including both low-energy gamma and X rays) as well as bremsstrahlung photons, the degree of underestimation would be even greater.

In most cases, the inventory of alpha-emitting radionuclides was determined by applying the 1-MeV photon conversion factor. Depending upon a number of factors, the extent of underestimation of the activity may be quite significant. In some cases, the alpha emitter may also emit gamma rays of relatively high energy. In other cases, low-energy gammas, X rays, or beta particles are emitted. In most cases, the alpha-emitting radionuclides have radioactive daughters which in turn may decay by emission of measurable radiation. Some level of uncertainty is associated with the degree of equilibrium which exists. If it can be assumed that a sufficient period of time has elapsed since production of the source to establish some degree of equilibrium, the daughter's photon emissions can be used to estimate the source activity of the parent. If the daughter decays by beta emission, the ensuing bremsstrahlung photons can also be used. A significant limiting factor for the alpha emitters is the sensitivity of the portable survey instruments used to make the measurements, as well as the background radiation present in many of the ORNL facilities. The typical ion chamber has a sensitivity of 0.1 mR/h. However, using this value along with the radionuclide's true dose-rate-to-curie conversion factor may result in calculated quantities of the radionuclide which are far from insignificant given their relatively low concentration limits. In addition, the radiation background at many facilities is at the very least a few tenths of a mR/h if not higher. At these values, any waste package surveys taken within a facility may represent background levels, in which case there is no activity present in the package being surveyed. It is also quite possible that any low-level radiation measurements are not caused by the alpha emitter in the waste package, but by insignificant levels of photon-emitting contaminants in the package. For all of these reasons, the actual activity of the alpha-emitting material calculated for this uncertainty analysis could then be significantly lower. The calculated most probable value assumes that the exposure rate reported for the package is entirely due to the alpha-emitting radionuclide; thus the activity is quite conservative.

Consideration of the significance of the disposal of the calculated quantity of the radionuclide may also permit the application of some "reasonableness" criterion to estimate the uncertainty of the calculated activity. The total activity of a transuranic (TRU) radionuclide calculated using its specific dose conversion factor may lead to an unreasonable activity, one which obviously is not physically possible. For the case of the transuranic radionuclides, a limiting factor (i.e., upper bound) was

applied to the package, such that, based upon the weight of the package contents, the total activity would not exceed 10 nCi/g. This value is 10% of the limit of 100 nCi/g, above which waste must be classified as TRU waste and may not be disposed of in SWSA 6. The value of 10% was chosen instead of the limit itself because an evaluation of all wastes disposed of in SWSA 6, using quite conservative assumptions, showed that essentially all waste packages were well under the limit. In fact, almost all were under 10 nCi/g. The use of 10 nCi/g is then considered to be a conservative upper bound for TRU activities. For those cases in which the disposal activity of fissile materials is determined from the UCN2681 form, the uncertainty would be rather low, especially relative to the other methods outlined above. Activities in these cases are determined either directly or indirectly from analytical measurements of the source material and are known to be within the uncertainty of the measurement.

Strontium 90 (and its daughter ^{90}Y) is a very significant contributor to the SWSA 6 inventory. This radionuclide emits only beta particles. However, the same 1-MeV-photon conversion factor is used to determine its activity in waste packages, which significantly underestimates the activity of ^{90}Sr . Fortunately, ^{90}Sr (and its daughter ^{90}Y) does emit relatively high-energy beta particles so that enough bremsstrahlung photons are produced in many waste packages to allow a measurable exposure rate from a package of pure ^{90}Sr . For the case of components contaminated with ^{90}Sr , the fraction of the incident beta energy converted into photons is only 2%. For other materials, the production of bremsstrahlung photons in that material varies with the energy of the beta particle and the atomic number of the absorber. The bremsstrahlung photons from ^{90}Sr exhibit a continuous spectrum with the peak energy flux occurring around 0.15 MeV. A similar distribution is produced for bremsstrahlung photons from radionuclides with lower-energy beta particles, but with a lower-energy peak. The exposure rate significance varies as a function of the same parameters as mentioned in the prior paragraph, with source and package playing an even greater role. Relative to a photon emitter such as ^{137}Cs , the conversion factor for ^{90}Sr in a lard can is 40 times lower, while in a 55-gal drum it is about 50 times smaller. Compared with the 1-MeV photon emitter upon which the ORNL conversion factor is based, the ^{90}Sr activity would be underestimated by even greater factors. A situation typically encountered is that in which the facility considers both ^{137}Cs and ^{90}Sr as the principal radionuclides and believes that each is present in equal quantities. Upon determining the curie content using the ORNL conversion factor, one-half of the total is assigned to each radionuclide. The error in this approach is that essentially all of the measured exposure rate is due to the ^{137}Cs , while little is due to the ^{90}Sr . The total activity calculated via the conversion factor should be attributed to ^{137}Cs , but, by dividing this activity in half, the facility introduces a factor of 2 underestimation in both the ^{137}Cs and ^{90}Sr activities. A fortuitous aspect of this assumption is that by assuming that ^{90}Sr is present in equal quantities with ^{137}Cs (as is assumed in almost all cases), the error in the ^{90}Sr activity is still much less than if ^{90}Sr alone was present.

In most cases, waste generators estimated the activity of certain very low-energy pure beta emitters, in particular ^3H and ^{14}C , by some means that they considered to be appropriate to the specific situation at hand (i.e., they did not use the 1-MeV point-source conversion factor). These estimates were taken as accurate but with some significant level of uncertainty. However, in some cases, activity for these radionuclides was calculated using the measured exposure rates and the

standard ORNL conversion factor. With the very low energy of the beta particle emitted, an appreciable quantity of the radionuclide is required to give any detectable exposure rate due to bremsstrahlung. The degree of underestimation of the reported activity is certainly quite significant but is difficult to quantify. In these cases, the most probable value for the activity in this uncertainty analysis was estimated to be 50 to 1000 times greater than that reported. This order of magnitude correction was somewhat arbitrarily chosen, but was empirically based upon the data available for the abovementioned estimated ^3H and ^{14}C activities.

A.4.7 Uncertainty Calculations for Activity Estimates

A.4.7.1 Deterministic Model Calculations

Calculations were performed using a deterministic model to compute the source activity and values for the low- and high-activity estimates. The results from the calculations were then used to check the reasonableness of the probabilistic model calculations. The deterministic model is based upon the following equation:

$$A = (D/R) \times C \quad (\text{A.2})$$

where

- A = estimated source activity,
- D = measured exposure rate at distance x from the source,
- R = expected exposure-rate-per-curie conversion factor for a material of density r at distance x from the source,
- C = distance correction factor for the error in the source-to-detector distance x.

The parameter D is nothing more than the actual value of the exposure rate measurement. R is the exposure-rate-to-curie conversion factor and is the most complex aspect of the calculation. It is radionuclide dependent, and is a function of the type of radiation emitted in the decay, the source geometry, composition, and density, and the distance from the source to the point of the exposure rate measurement. The exposure-rate-to-curie conversion factor has been used for all radionuclides of interest, including those that emit high- or low-energy gamma rays, X rays, or beta particles (with the ensuing bremsstrahlung photons). This group includes any radionuclides that have radioactive daughters that may emit such radiation, which is particularly important for some of the transuranics. The parameter C is one that has been added to reflect the effect on the exposure rate caused by an error in the source-to-detector distance.

To describe the methodology used, a simple example calculation is provided. The basis for the assumptions made are pointed out as they appear in the calculation. The example calculation consists of a waste package (the "5-gal" lard can) from Building 3525. The principal nuclide is ^{60}Co .

Although the exposure rate used in the activity determination was 648 R/h, the actual measured exposure rate was 72 R/h, taken at approximately 3 ft using a portable ion chamber. Facility personnel used an inverse square relationship to derive a 1-ft exposure rate of 648 R/h that was used in the activity calculation with the standard 1-MeV-photon conversion factor. Uncertainties are associated with the measured exposure rate due to calibration tolerances and the energy dependence of the instrument response. The response of the standard ORNL ion chamber is fairly linear over a wide range of energies, hence the overall uncertainty attributed to the measured exposure rate was assumed to be 10%.

For this example, shielding calculations were performed using a cylindrical source (for the lard can) of ^{60}Co with dimensions of 10-in. (diam) by 14 in. (height). An iron shield (0.06-cm thickness) was included in the calculations to approximate the attenuation effects of the steel lard can wall. The source material density was varied from 0.5 to 2.0 g/cm³ in increments of 0.5 g/cm³. Because the density of iron is 7.86 g/cm³, a lard can filled completely with 2.0-g/cm³- density material corresponds to a lard can with 25 vol % steel (iron) and 75-vol % void space (this void space might contain a very small volume of low-density material such as paper and plastic waste). The weight of such a can would be approximately 100 lbs. Since the reported weight for the waste packages ranged from 20 to 100 lb, the use of 2.0 g/cm³ as the upper bound for the average waste material density is appropriate. The lower bound for density of the waste material was 0.5 g/cm³. This value translates to a 20 lb lard can completely filled with this material. This value is deemed to be reasonable in terms of a lower bound. The exposure rate at a distance of 3 ft was calculated for a 1 Ci source of ^{60}Co , which represents an exposure-rate-to-curie conversion factor for this distance as a functions of source density and geometry. At 3 ft, the exposure rate for material densities of 0.5 g/cm³ and 2.0 g/cm³ ranged from approximately 1.4 to 1.0 R/h per curie respectively.

Exposure rates as a function of distance were also calculated for the cylindrical source used above with a density of 1.0 g/cm³. This calculation was done to examine the effect on the exposure rate, since there is uncertainty associated with the distance at which the measurement was obtained. For this example, an uncertainty of 1 ft was considered to be reasonable, taking into consideration the manner in which the measurement was made. The calculations showed that the exposure rate at a source-to-detector distance of 2 ft was 2.5 times greater than the response obtained at 3 ft. Conversely, the exposure rate response at 4 ft was 0.6 times that obtained at 3 ft. At these distances, the lard can approximates a point source reasonably well.

Even though in the above example the exposure rate was obtained with a portable ion chamber, many facilities use in-cell probes to obtain exposure rate measurements on waste packages. The uncertainty associated with these detectors is undoubtedly greater than that for the portable ion chamber, perhaps in the range of 25%.

A.4.7.2 Source-Activity Estimate

In the example above, an exposure rate of 72 R/h was measured at a distance of 3 ft. The source activity estimate is determined as follows:

$$A = (72/1.2) = 60 \text{ Ci}, \quad (\text{A.3})$$

where

72 R/h = measured exposure rate at 3 ft,

1.2 R/h/Ci = average exposure rate per curie (for $r = 0.5$ and 2.0 g/cm^3).

Since the source activity estimate is based upon the measurement at 3 ft, there is no distance correction factor (i.e., $C = 1.0$).

A.4.7.3 Low-Activity Estimate

In order to estimate the lower bound of the activity, the values of D and C should be minimized, while R should be maximized. Substituting into the above equation gives:

$$A = (64.8/1.4) \times 0.6 = 27.7 \approx 28 \text{ Ci}, \quad (\text{A.4})$$

where

64.8 R/h = lower bound of the measured exposure rate at 3 ft (90% of 72 R/h),

1.4 R/h/Ci = exposure rate per curie for $r = 0.5 \text{ g/cm}^3$,

0.6 = distance correction factor for x of 4 ft.

A.4.7.4 High-Activity Estimate

In order to estimate the upper bound of the activity, the values of D and C should be maximized, while R should be minimized. Substituting into the above equation gives:

$$A = (79.2/1.0) \times 2.5 = 198 \approx 200 \text{ Ci}, \quad (\text{A.5})$$

where

79.2 R/h = upper bound of the measured exposure rate at 3 ft (110% of 72 R/h),

1.0 R/h/Ci = exposure rate per curie for $r = 2.0 \text{ g/cm}^3$,

2.5 = distance correction factor for x of 2 ft.

A.4.7.5 Deterministic Model Conclusions

As can be seen, this deterministic model gives an estimated activity for ^{60}Co in the waste package of 60 Ci, with a range of 28 to 200 Ci. The activity as determined on the UCN16114 for this package (using the ORNL 1-MeV photon conversion factor) was 110 Ci. The activity estimate using the 6 C E n formula for ^{60}Co (with a exposure rate of 648 R/h exposure rate at 1 ft, 1.25 MeV energy and 2 photons per disintegration) is 43 Ci.

The deterministic values for the source activity calculated above provide the estimated source activity and range, but do not allow for the independent variability of the uncertainties associated with the input parameters, such as the ion chamber exposure rate response, the exposure-rate conversion factor associated with the waste material density, and the distance at which the measurement was taken. This approach gives the range of source activities that are possible, but does not tell what is probable.

A.4.7.6 Probabilistic Model Conclusions

The most probable source activity and its associated uncertainty for each radionuclide was calculated using Crystal Ball® for Windows™ by propagating the uncertain parameters through a model of the joint distribution to produce a distribution of activity predictions. Latin Hypercube Sampling (Iman and Shortencarrier 1984) was used as the numerical method for propagation. Typically, Crystal Ball® uses Monte Carlo sampling, generating random numbers for a probability distribution over the entire range of possible values for that distribution. For this reason, a large number of trials is required to obtain results that approximate the true shape of the distribution. With Latin Hypercube sampling, an assumption's probability distribution is divided into intervals of equal probability. Crystal Ball® then generates an assumption value for each interval according to the interval's probability distribution. Compared with conventional Monte Carlo sampling, Latin Hypercube sampling is more precise because the entire range of the distribution is sampled in a more even, consistent manner.

To account for the uncertainties in the input parameters, a numerical method using Latin Hypercube Sampling was used to determine a probability distribution for the source activity. For the Building 3535 ^{60}Co example, three input parameters, consisting of the measured exposure rate at 3 ft, the exposure-rate-per-curie response at 3 ft, and the distance correction factor, were used. The uncertainties associated with these parameters were propagated. A normal distribution with a mean of 72 R/h and a standard deviation of 2.4 R/h was used to describe the probability distribution for the measured exposure rate. A uniform probability distribution with a range of 1.0 to 1.4 R/h/Ci was used for the exposure-rate-per-curie response. The distance probability distribution was described using a distance correction factor that was distributed normally with a mean of 3.00 ft and a standard deviation of 0.34 ft. The parametric information for the input parameters are shown in Fig. A.2

A run with 1000 iterations was made for this example: This calculation allows for the determination of a ^{60}Co activity probability distribution, with a most probable activity, and an upper-

Assumption: Measured exposure rate

Normal distribution with parameters:

Mean

72.00

Standard deviation

2.40

Assumption: Exposure rate per Curie

Uniform distribution with parameters:

Minimum

1.00

Maximum

1.40

Assumption: Distance

Normal distribution with parameters:

Mean

3.00

Standard deviation

0.34

Fig. A.2. Input parameter uncertainty for the ^{60}Co example.

and lower-bound value, each with an associated confidence interval. The most probable activity (i.e., the mode of the calculated activity probability distribution) was 54.4 Ci, with a range minimum at 95% confidence of 31.9 Ci and a range maximum at 95% confidence of 92.1 Ci. The 95% confidence interval represents 1.96 standard deviations about the mean of the calculated activity probability distribution. Recall that the deterministic model gave an estimated activity of ^{60}Co in the waste of 60 Ci, with a range of 28 to 200 Ci. The activity as determined on the UCN16114 for this package (using the ORNL 1-MeV photon conversion factor) was 110 Ci. The probability distribution resulting from the Latin Hypercube Sampling forecast is shown in Fig. A.3.

Similar uncertainty analyses were performed for other radionuclides and source geometries. An additional uncertainty distribution was introduced into the calculation whenever it was assumed that a package contained equal quantities of two or more radionuclides. In most cases, this was for packages containing ^{137}Cs and ^{90}Sr . The uncertainty represents the extent to which the radionuclides are present in equal quantities. For most cases, the fraction of the total represented by ^{137}Cs was set equal to 0.5, and was represented by a uniform probability distribution with a low of 0.2 and a high of 0.8. The ^{90}Sr fraction is then 1 minus the ^{137}Cs fraction. The addition of this uncertainty results in a larger range in activity (from the minimum to the maximum activity estimates) than would be observed for each individual radionuclide.

This probabilistic approach facilitates the estimation of source activity for packages of waste based upon the principal radionuclide(s) identified, the measured exposure rate at the specified distance, the instrument used for the measurement, the exposure-rate-per-curie response for the specified distance (a function of source density and geometry), and, if present, the mixture of multiple radionuclides in a package.

A.4.8 Shielding Calculations

The exposure rate response calculations were performed using MicroShield™, a product of Grove Engineering, Inc., and a personal computer (PC) version of the shielding code, ISOSHL. The latter program uses point kernel integration (with buildup) to calculate the radiation exposure rate at a detector point for bremsstrahlung and/or decay gamma rays emitted by radionuclide sources. For each waste package type considered, information about the geometry of the problem, radionuclides in the source, and the material composition of the source and shield regions were used as input to MicroShield.™ The MicroShield™ code libraries contain the information used to calculate the exposure rate and radiation spectra, (i.e., attenuation coefficients, gamma ray decay energies and probabilities, maximum beta energies and probabilities, and buildup factor data). All shielding calculations that were performed to determine the detector response to the waste packages in which the waste material was disposed used a cylindrical source and, if appropriate, with a cylindrical shield to account for the wall of the waste package. For the low-activity waste that was typically disposed of in small plastic bags, there would be no shield.

Fig. A.3. Probability distribution of ^{60}Co source activity

A.5 NON-IWMF INVENTORY

A.5.1 Key Radionuclides

The non-IWMF key radionuclides were selected based on a groundwater analysis of the inventories presented in Tables A.2 to A.13. The radionuclides that came within a factor of 1000 of the groundwater limit were initially chosen. This analysis resulted in the following list of key radionuclides:

^{14}C , ^{99}Tc , ^{232}Th , ^{233}U , ^{238}U , ^{239}Pu , ^{241}Am .

The key nuclides from the 1994 PA were:

^3H , ^{14}C , ^{60}Co , ^{90}Sr , ^{99}Tc , ^{137}Cs , ^{238}U .

Combining these two lists results in:

^3H , ^{14}C , ^{60}Co , ^{90}Sr , ^{99}Tc , ^{137}Cs , ^{232}Th , ^{233}U , ^{238}U , ^{239}Pu , ^{241}Am .

Cobalt 60 was dropped from this list for three reasons: (1) it has a relatively short half-life, (2) it is not a problem in the non-IWMF facilities based on calculations for groundwater and surface water, and (3) the disposal capacity for ^{60}Co in the IWMF is very large. Thus the final key radionuclides were:

The location of key radionuclides present in the SWSA 6 non-IWMF disposal facilities are presented in Table A.15.

A.5.2 Uncertainty in the Non-IWMF Inventory

Uncertainties in the non-IWMF radionuclide inventory were estimated to support uncertainty analyses for the revision of the 1994 PA. Since the 1994 PA, several changes to the non-IWMF part of SWSA 6 had occurred: (1) the number of disposal facilities under each disposal facility type accounted for in the 1994 PA had changed, (2) new disposal facility types had been added, and (3) the radionuclide inventory had changed in some cases.

To account for these changes, the following steps were used in estimating the uncertainties associated with the radionuclide inventories:

1. Applied the uncertainties in the radionuclide inventory to the reported inventories instead of the most probable activity (MPA). The 1994 PA modeling was based on the MPA values and its associated uncertainty.
2. Used the same magnitude of uncertainty from the 1994 PA for cases where the inventory had not changed appreciably (within approximately +/- 20%).
3. Used the largest (most conservative) magnitude of uncertainty used for the radionuclide from the 1994 PA for cases where the inventory had changed appreciably or the radionuclide was not considered or did not exist in the inventory of the 1994 PA. [Changes to the inventory (from the PA standpoint) could be due to several reasons. Since the last PA, waste packages had been added or removed from certain disposal units. New disposal facility types have been added (i.e., the Quadrex trenches). Use of "zones" has resulted in a redistribution of the inventory for calculational purposes. Review of inventory records resulted in an update of the WTS database.]
4. Computed a relative most probable activity for each radionuclide by multiplying the MPAs with the ratio between the 1994 inventory and the current reported inventory. If no MPAs values were available, the relative MPAs were simply the reported inventory multiplied by 10.
5. Used the relative MPA as the upper or lower bound for cases where the upper or lower bound for the uncertainty (associated with the activity for each radionuclide) did not encompass the relative MPA value.

Table A.15 Key radionuclides in SWSA 6 non-IWMF disposal facilities

Disposal Facility	Key Radionuclides Present									
High-range silo Zone 3	³ H	¹⁴ C	⁹⁰ Sr		¹³⁷ Cs	²³² Th		²³⁸ U		
High-range wells Zone 5			⁹⁰ Sr		¹³⁷ Cs					
High-range wells-in-silos Zone 5			⁹⁰ Sr	⁹⁹ Tc	¹³⁷ Cs	²³² Th				
Fissile wells Zone 7					¹³⁷ Cs	²³² Th		²³⁸ U		
Quadrex trenches Zone 8			⁹⁰ Sr		¹³⁷ Cs	²³² Th		²³⁸ U		
Low-range silos Zone 8	³ H	¹⁴ C	⁹⁰ Sr	⁹⁹ Tc	¹³⁷ Cs	²³² Th	²³³ U	²³⁸ U	²³⁹ Pu	²⁴¹ Am
Biological trenches Zone 15			⁹⁰ Sr		¹³⁷ Cs					
Asbestos silos Zone 16			⁹⁰ Sr	⁹⁹ Tc	¹³⁷ Cs	²³² Th		²³⁸ U	²³⁹ Pu	²⁴¹ Am
Biological trenches Zone 20			⁹⁰ Sr		¹³⁷ Cs					
Tumulus II Zone 26	³ H	¹⁴ C	⁹⁰ Sr		¹³⁷ Cs	²³² Th	²³³ U	²³⁸ U	²³⁹ Pu	²⁴¹ Am
Tumulus I Zone 27	³ H	¹⁴ C	⁹⁰ Sr	⁹⁹ Tc	¹³⁷ Cs	²³² Th	²³³ U	²³⁸ U	²³⁹ Pu	²⁴¹ Am
Low-range silos Zone 30	³ H	¹⁴ C	⁹⁰ S		¹³⁷ Cs	²³² Th	²³³ U	²³⁸ U	²³⁹ Pu	²⁴¹ Am

The results for the uncertainties associated with the non-IWMF key radionuclide inventory are presented in Table A.16 through Table A.27. These tables provide the mode, minimum, and maximum amounts (in grams) of key radionuclides for each facility. The first column contains the list of key radionuclides. The second column contains the reported inventory in g/facility. The third column contains the distribution type assigned to the radionuclide inventory data. The fourth and fifth columns contain the minimum and maximum inventory. These columns were derived by taking the reported inventories and applying the magnitude of uncertainty ("spread") determined in the 1994 PA. The inventory uncertainty in the 1994 PA was applied with respect to the MPAs. Thus, the ratios of the maximum and minimum activities with respect to the MPAs (from 1994) were multiplied with the current reported inventory to obtain most of the values in columns four and five. (since the current reported inventory is used in this PA instead of the MPAs). The ratio multiplication method was not used wherever significant changes to the inventories, since 1994, were encountered. In these

Table A.16 Radionuclide inventory uncertainties for the high-range silos (Zone 3).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
³ H	2.88×10^{-5}	Triangular	1.44×10^{-5}	4.32×10^{-5}	1.00×10^1	2.87×10^{-5}	1.00	2.87×10^{-5}	2.88×10^{-5}	1.44×10^{-5}	4.32×10^{-5}
¹⁴ C	2.56×10^{-4}	Triangular	1.28×10^{-4}	3.84×10^{-4}	4.00×10^{-2}	2.49×10^{-4}	0.97	2.42×10^{-4}	2.56×10^{-4}	1.28×10^{-4}	3.84×10^{-4}
⁹⁰ Sr	2.76×10^{-3}	Triangular	1.83×10^{-3}	4.69×10^{-3}	3.51×10^2	6.99×10^{-2}	1.67	1.17×10^{-1}	2.76×10^{-3}	1.83×10^{-3}	1.17×10^{-1}
¹³⁷ Cs	5.16×10^{-3}	Triangular	3.10×10^{-3}	7.22×10^{-3}	2.57×10^1	8.21×10^{-3}	1.59	1.31×10^{-2}	5.16×10^{-3}	3.10×10^{-3}	1.31×10^{-2}
²³² Th	3.43×10^1	Triangular	3.09×10^1	3.77×10^1	1.36×10^{-4}	3.44×10^1	1.00	3.46×10^1	3.43×10^1	3.09×10^1	3.77×10^1
²³⁸ U	8.24×10^2	Triangular	7.42×10^2	9.06×10^2	1.10×10^{-2}	9.08×10^2	1.00	9.06×10^2	8.24×10^2	7.42×10^2	9.06×10^2
²³⁹ Pu	4.60×10^{-4}	Triangular	4.60×10^{-5}	4.60×10^{-3}	1.50×10^{-2}	6.72×10^{-3}	0.98	6.61×10^{-3}	4.60×10^{-4}	4.60×10^{-5}	6.61×10^{-3}

^aInventory based on 36 silos.

Table A.17 Radionuclide inventory uncertainties for the high-range wells (Zone 5).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
⁹⁰ Sr	1.09×10^0	Triangular	7.30×10^{-1}	1.45×10^0	1.36×10^3	1.08×10^0	0.99	1.08×10^0	1.09×10^0	7.30×10^{-1}	1.45×10^0
¹³⁷ Cs	1.70×10^0	Triangular	1.53×10^0	1.96×10^0	1.40×10^3	1.79×10^0	1.05	1.88×10^0	1.70×10^0	1.53×10^0	1.96×10^0

^aInventory based on 9 wells.

Table A.18 Radionuclide inventory uncertainties for the high-range wells-in-silos (Zone 5).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
⁹⁰ Sr	3.12×10^{-2}	Triangular	1.97×10^{-3}	7.68×10^{-2}	NA	NA	NA	3.12×10^{-2}	3.12×10^{-2}	1.97×10^{-3}	7.68×10^{-2}
⁹⁹ Tc	4.80×10^{-1}	Triangular	4.56×10^{-1}	5.04×10^{-1}	NA	NA	NA	4.80×10^0	4.80×10^{-1}	4.56×10^{-1}	4.80×10^0
¹³⁷ Cs	2.58×10^{-1}	Triangular	1.55×10^{-1}	3.61×10^{-1}	NA	NA	NA	2.58×10^0	2.58×10^{-1}	1.55×10^{-1}	2.58×10^0
²³² Th	6.92×10^0	Triangular	6.23×10^0	7.61×10^0	NA	NA	NA	6.92×10^1	6.92×10^0	6.23×10^0	6.92×10^1

^aInventory based on 49 wells-in-silos.**Table A.19 Radionuclide inventory uncertainties for the fissile wells (Zone 7).**

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
¹³⁷ Cs	2.44×10^{-1}	Triangular	1.46×10^{-1}	3.42×10^{-1}	4.25×10^1	2.44×10^{-1}	1	2.44×10^0	2.44×10^{-1}	1.46×10^{-1}	2.44×10^0
²³² Th	5.50×10^0	Triangular	3.85×10^0	6.38×10^0	NA	NA	NA	5.50×10^1	5.50×10^0	3.85×10^0	5.50×10^1
²³⁵ U	1.36×10^2	Triangular	1.36×10^1	1.36×10^3	8.75×10^{-4}	2.02×10^2	1.49	3.01×10^2	1.36×10^2	1.36×10^1	1.36×10^3
²³⁸ U	7.28×10^3	Triangular	6.55×10^3	8.01×10^3	4.83×10^{-3}	7.18×10^3	0.99	7.08×10^3	7.28×10^3	6.55×10^3	8.01×10^3

^aInventory based on 2 wells.

Table A.20 Radionuclide inventory uncertainties for the Quadrex trenches (Zone 8).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
⁹⁰ Sr	7.81×10^{-7}	Triangular	4.92×10^{-8}	1.92×10^{-6}	NA	NA	NA	7.81×10^{-6}	7.81×10^{-7}	4.92×10^{-8}	7.81×10^{-6}
¹³⁷ Cs	1.87×10^{-3}	Triangular	1.12×10^{-3}	2.62×10^{-3}	NA	NA	NA	1.87×10^{-2}	1.87×10^{-3}	1.12×10^{-3}	1.87×10^{-2}
²³² Th	2.84×10^3	Triangular	1.99×10^3	3.29×10^3	NA	NA	NA	2.84×10^4	2.84×10^3	1.99×10^3	2.84×10^4
²³⁵ U	1.97×10^1	Triangular	1.97×10^0	1.97×10^2	NA	NA	NA	1.97×10^2	1.97×10^1	1.97×10^0	1.97×10^2
²³⁸ U	9.30×10^2	Triangular	7.10×10^2	1.28×10^3	NA	NA	NA	9.30×10^3	9.30×10^2	7.10×10^2	9.30×10^3

^aInventory based on 4 trenches.

Table A.21 Radionuclide inventory uncertainties for the low-range silos (Zone 8).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
³ H	7.30×10^{-5}	Triangular	3.65×10^{-5}	1.10×10^{-4}	1.02×10^1	NA	NA	7.30×10^{-4}	7.30×10^{-5}	3.65×10^{-5}	7.30×10^{-4}
¹⁴ C	5.80×10^{-5}	Triangular	2.90×10^{-5}	8.70×10^{-5}	7.23×10^1	NA	NA	5.80×10^{-4}	5.80×10^{-5}	2.90×10^{-5}	5.80×10^{-4}
⁹⁰ Sr	5.76×10^{-3}	Triangular	3.63×10^{-4}	1.42×10^{-2}	5.34	NA	NA	5.76×10^{-2}	5.76×10^{-3}	3.63×10^{-4}	5.76×10^{-2}
⁹⁹ Tc	1.41×10^{-1}	Triangular	1.41×10^{-2}	1.41×10^0	3.26	NA	NA	1.41×10^0	1.41×10^{-1}	1.41×10^{-2}	1.41×10^0
¹³⁷ Cs	7.03×10^{-3}	Triangular	4.22×10^{-3}	9.84×10^{-3}	3.23	NA	NA	7.03×10^{-2}	7.03×10^{-3}	4.22×10^{-3}	7.03×10^{-2}
²³² Th	2.63×10^3	Triangular	1.84×10^3	3.05×10^3	1.08×10^{-3}	NA	NA	2.63×10^4	2.63×10^3	1.84×10^3	2.63×10^4
²³³ U	4.18×10^{-2}	Triangular	4.18×10^{-3}	4.18×10^{-1}	2.75×10^{-2}	NA	NA	4.18×10^{-1}	4.18×10^{-2}	4.18×10^{-3}	4.18×10^{-1}
²³⁵ U	1.60×10^1	Triangular	1.44×10^1	1.76×10^1	NA	NA	NA	1.60×10^2	1.60×10^1	1.44×10^1	1.60×10^2
²³⁸ U	2.00×10^3	Triangular	1.53×10^3	2.75×10^3	1.37×10^{-1}	NA	NA	2.00×10^4	2.00×10^3	1.53×10^3	2.00×10^4
²³⁹ Pu	7.30×10^{-2}	Triangular	7.30×10^{-3}	7.30×10^{-1}	3.47×10^{-1}	NA	NA	7.30×10^{-1}	7.30×10^{-2}	7.30×10^{-3}	7.30×10^{-1}
²⁴¹ Am	1.19×10^{-3}	Triangular	1.19×10^{-4}	1.19×10^{-2}	2.76×10^{-2}	NA	NA	1.19×10^{-2}	1.19×10^{-3}	1.19×10^{-4}	1.19×10^{-2}
²⁴³ Am	5.68×10^{-5}	Triangular	5.68×10^{-6}	5.68×10^{-4}	1.08×10^{-2}	NA	NA	5.68×10^{-4}	5.68×10^{-5}	5.68×10^{-6}	5.68×10^{-4}

^aInventory based on 24 silos.

Table A.22 Radionuclide inventory uncertainties for the biological trenches (Zone 15).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
⁹⁰ Sr	2.23×10^{-5}	Triangular	1.56×10^{-5}	2.90×10^{-5}	1.81×10^{-1}	NA	NA	2.23×10^{-4}	2.23×10^{-5}	1.56×10^{-5}	2.23×10^{-4}
¹³⁷ Cs	8.23×10^{-6}	Triangular	6.58×10^{-6}	1.07×10^{-5}	2.25×10^{-3}	6.47×10^{-6}	0.79	5.08×10^{-6}	8.23×10^{-6}	5.08×10^{-6}	1.07×10^{-5}

^aInventory based on 4 trenches.

Table A.23 Radionuclide inventory uncertainties for the asbestos silos (Zone 16).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
¹⁴ C	3.45×10^{-5}	Triangular	1.73×10^{-5}	5.18×10^{-5}	5.79×10^{-1}	1.18×10^{-2}	3.42	4.04×10^{-2}	3.45×10^{-5}	1.73×10^{-5}	4.04×10^{-2}
⁹⁰ Sr	1.86×10^{-5}	Triangular	4.84×10^{-6}	4.05×10^{-5}	4.71×10^{-2}	3.07×10^{-5}	1.49	4.56×10^{-5}	1.86×10^{-5}	4.84×10^{-6}	4.56×10^{-5}
⁹⁹ Tc	7.33×10^{-3}	Triangular	3.96×10^{-4}	1.97×10^{-2}	1.02×10^{-3}	5.47×10^{-3}	0.88	4.81×10^{-3}	7.33×10^{-3}	3.96×10^{-4}	1.97×10^{-2}
¹³⁷ Cs	8.75×10^{-5}	Triangular	7.00×10^{-5}	1.14×10^{-4}	5.67×10^{-2}	5.93×10^{-5}	0.68	4.02×10^{-5}	8.75×10^{-5}	4.02×10^{-5}	1.14×10^{-4}
²³² Th	6.95×10^1	Triangular	4.87×10^1	8.06×10^1	NA	NA	NA	6.95×10^2	6.95×10^1	4.87×10^1	6.95×10^2
²³⁸ U	1.30×10^2	Triangular	9.67×10^1	1.79×10^2	5.08×10^{-1}	1.37×10^1	0.56	7.73×10^2	1.30×10^2	9.67×10^1	7.73×10^2
²³⁹ Pu	1.81×10^{-2}	Triangular	1.81×10^{-3}	1.81×10^{-1}	NA	NA	NA	1.81×10^{-1}	1.81×10^{-2}	1.81×10^{-3}	1.81×10^{-1}
²⁴¹ Am	4.51×10^{-6}	Triangular	4.51×10^{-7}	4.51×10^{-5}	NA	NA	NA	4.51×10^{-5}	4.51×10^{-6}	4.51×10^{-7}	4.51×10^{-5}

^aInventory based on 11 silos.

Table A.24 Radionuclide inventory uncertainties for the biological trenches (Zone 20).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
⁹⁰ Sr	2.16×10^{-5}	Triangular	1.51×10^{-5}	2.81×10^{-5}	1.81×10^{-1}	NA	NA	2.16×10^{-4}	2.16×10^{-5}	1.51×10^{-5}	2.16×10^{-4}

^aInventory based on 2 trenches.**Table A.25 Radionuclide inventory uncertainties for Tumulus II (Zone 26).**

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
³ H	7.15×10^{-7}	Triangular	3.58×10^{-7}	1.07×10^{-6}	1.61	7.60×10^{-7}	1.00	7.60×10^{-7}	7.15×10^{-7}	3.58×10^{-7}	1.07×10^{-6}
¹⁴ C	4.34×10^{-6}	Triangular	2.17×10^{-6}	6.51×10^{-6}	2.12	2.17×10^{-3}	1.00	2.17×10^{-3}	4.34×10^{-6}	2.17×10^{-6}	2.17×10^{-3}
⁹⁰ Sr	9.43×10^{-5}	Triangular	2.45×10^{-5}	2.06×10^{-4}	3.29	1.08×10^{-4}	2.76	2.98×10^{-4}	9.43×10^{-5}	2.45×10^{-5}	2.98×10^{-4}
¹³⁷ Cs	4.02×10^{-4}	Triangular	3.22×10^{-4}	5.23×10^{-4}	12.9	6.77×10^{-4}	1.69	1.14×10^{-3}	4.02×10^{-4}	3.22×10^{-4}	1.14×10^{-3}
²³² Th	2.08×10^1	Triangular	1.46×10^1	2.41×10^1	3.97×10^{-4}	1.65×10^1	1.00	1.66×10^1	2.08×10^1	1.46×10^1	2.41×10^1
²³³ U	1.97×10^{-2}	Triangular	1.97×10^{-3}	1.97×10^{-1}	5.19×10^{-2}	2.46×10^{-2}	1.00	2.45×10^{-2}	1.97×10^{-2}	1.97×10^{-3}	1.97×10^{-1}
²³⁸ U	4.74×10^1	Triangular	3.62×10^1	6.52×10^1	7.33×10^{-3}	9.95×10^1	0.33	3.25×10^1	4.74×10^1	3.25×10^1	6.52×10^1
²³⁹ Pu	6.28×10^{-4}	Triangular	6.28×10^{-5}	6.28×10^{-3}	5.21×10^{-2}	3.84×10^{-3}	1.24	4.77×10^{-3}	6.28×10^{-4}	6.28×10^{-5}	6.28×10^{-3}
²⁴¹ Am	1.66×10^{-5}	Triangular	1.66×10^{-6}	1.66×10^{-4}	1.05×10^{-1}	1.40×10^{-4}	1.00	1.40×10^{-4}	1.66×10^{-5}	1.66×10^{-6}	1.66×10^{-4}

^aInventory based on 219 vaults.

Table A.26 Radionuclide inventory uncertainties for Tumulus I (Zone 27).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA per facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
³ H	9.61×10^{-7}	Triangular	4.81×10^{-7}	1.44×10^{-6}	2.7	9.60×10^{-7}	1.00	9.60×10^{-7}	9.61×10^{-7}	4.81×10^{-7}	1.44×10^{-6}
¹⁴ C	2.95×10^{-5}	Triangular	1.48×10^{-5}	4.43×10^{-5}	9.76	7.52×10^{-3}	0.51	3.83×10^{-3}	2.95×10^{-5}	1.48×10^{-5}	3.83×10^{-3}
⁹⁰ Sr	3.03×10^{-4}	Triangular	1.91×10^{-5}	7.45×10^{-4}	3.71	9.14×10^{-5}	0.27	2.46×10^{-5}	3.03×10^{-4}	1.91×10^{-5}	7.45×10^{-4}
⁹⁹ Tc	1.48×10^{-2}	Triangular	1.48×10^{-3}	1.48×10^{-1}	3.69×10^{-1}	7.48×10^{-2}	0.03	2.40×10^{-3}	1.48×10^{-2}	1.48×10^{-3}	1.48×10^{-1}
¹³⁷ Cs	9.76×10^{-4}	Triangular	6.83×10^{-4}	1.13×10^{-3}	1.40×10^1	5.53×10^{-4}	0.57	3.14×10^{-4}	9.76×10^{-4}	3.14×10^{-4}	1.13×10^{-3}
²³² Th	1.94×10^1	Triangular	1.36×10^1	2.25×10^1	1.11×10^{-4}	3.48×10^0	0.27	9.32×10^{-1}	1.94×10^1	9.32×10^{-1}	2.25×10^1
²³³ U	1.73×10^{-2}	Triangular	1.73×10^{-3}	1.73×10^{-1}	9.81×10^{-2}	3.50×10^{-2}	0.21	7.35×10^{-3}	1.73×10^{-2}	1.73×10^{-3}	1.73×10^{-1}
²³⁸ U	7.80×10^1	Triangular	5.95×10^1	1.07×10^2	4.55×10^{-2}	4.65×10^2	0.30	1.39×10^2	7.80×10^1	5.95×10^1	1.39×10^2
²³⁷ Np	3.90×10^{-2}	Triangular	3.90×10^{-3}	3.90×10^{-1}	NA	NA	NA	3.90×10^{-1}	3.90×10^{-2}	3.90×10^{-3}	3.90×10^{-1}
²³⁹ Pu	2.82×10^{-3}	Triangular	2.82×10^{-4}	2.82×10^{-2}	9.03×10^{-2}	5.01×10^{-3}	0.23	1.15×10^{-3}	2.82×10^{-3}	2.82×10^{-4}	2.82×10^{-2}
²⁴⁰ Pu	1.02×10^{-4}	Triangular	1.02×10^{-5}	1.02×10^{-3}	0	0.00×10^0	1.01	1.02×10^{-3}	1.02×10^{-4}	1.02×10^{-5}	1.02×10^{-3}
²⁴¹ Am	6.07×10^{-5}	Triangular	6.07×10^{-6}	6.07×10^{-4}	8.27×10^{-2}	8.29×10^{-5}	0.37	3.05×10^{-5}	6.07×10^{-5}	6.07×10^{-6}	6.07×10^{-4}
²⁴³ Am	2.11×10^{-4}	Triangular	2.11×10^{-5}	2.11×10^{-3}	1.86×10^{-3}	3.21×10^{-5}	0.18	5.81×10^{-6}	2.11×10^{-4}	5.81×10^{-6}	2.11×10^{-3}

^aInventory based on 291 vaults.

Table A.27 Radionuclide inventory uncertainties for low-range silos (Zone 30).

Nuclide	Reported inventory ^a (g/facility)	Distribution type	1994 Minimum inventory (g/facility)	1994 Maximum inventory (g/facility)	94 MPA (Ci)	94 MPA pre facility (g/facility)	Ratio of 94 to current inventory	Relative 94 MPA per facility (g/facility)	Mode (g/facility)	Revised minimum inventory (g/facility)	Revised maximum inventory (g/facility)
³ H	9.52×10^{-6}	Triangular	4.76×10^{-6}	1.43×10^{-5}	1.02×10^1	NA	NA	9.52×10^{-5}	9.52×10^{-6}	4.76×10^{-6}	9.52×10^{-5}
¹⁴ C	1.48×10^{-4}	Triangular	7.40×10^{-5}	2.22×10^{-4}	7.23×10^1	NA	NA	1.48×10^{-3}	1.48×10^{-4}	7.40×10^{-5}	1.48×10^{-3}
⁹⁰ Sr	6.49×10^{-4}	Triangular	4.09×10^{-5}	1.60×10^{-3}	5.34	NA	NA	6.49×10^{-3}	6.49×10^{-4}	4.09×10^{-5}	6.49×10^{-3}
¹³⁷ Cs	1.36×10^{-3}	Triangular	8.16×10^{-4}	1.90×10^{-3}	3.23	NA	NA	1.36×10^{-2}	1.36×10^{-3}	8.16×10^{-4}	1.36×10^{-2}
²³² Th	3.28×10^2	Triangular	2.30×10^2	3.80×10^2	1.08×10^{-3}	NA	NA	3.28×10^3	3.28×10^2	2.30×10^2	3.28×10^3
²³³ U	6.05×10^{-2}	Triangular	6.05×10^{-3}	6.05×10^{-1}	2.75×10^{-2}	NA	NA	6.05×10^{-1}	6.05×10^{-2}	6.05×10^{-3}	6.05×10^{-1}
²³⁸ U	8.16×10^2	Triangular	6.23×10^2	1.12×10^3	NA	NA	NA	8.16×10^3	8.16×10^2	6.23×10^2	8.16×10^3
²³⁷ Np	1.58×10^{-2}	Triangular	1.58×10^{-3}	1.58×10^{-1}	NA	NA	NA	1.58×10^{-1}	1.58×10^{-2}	1.58×10^{-3}	1.58×10^{-1}
²³⁹ Pu	3.67×10^{-3}	Triangular	3.67×10^{-4}	3.67×10^{-2}	3.47×10^{-1}	NA	NA	3.67×10^{-2}	3.67×10^{-3}	3.67×10^{-4}	3.67×10^{-2}
²⁴¹ Am	6.58×10^{-5}	Triangular	6.58×10^{-6}	6.58×10^{-4}	2.76×10^{-2}	NA	NA	6.58×10^{-4}	6.58×10^{-5}	6.58×10^{-6}	6.58×10^{-4}
²⁴³ Am	7.66×10^{-4}	Triangular	7.66×10^{-5}	7.66×10^{-3}	1.08×10^{-2}	NA	NA	7.66×10^{-3}	7.66×10^{-4}	7.66×10^{-5}	7.66×10^{-3}

^aInventory based on 105 silos.

instances, the most conservative ratio (of maximum and minimum activities to the MPAs) used for the radionuclide in the 1994 PA was applied.

Once the minimum and maximum inventories based on the current reported inventories were determined, the next step involved making sure that the MPAs fell within the uncertainty range for that radionuclide. Since the inventories had changed in some cases, a "relative MPA" was computed. This computation was done by multiplying the the MPA (column 8) with the ratio between the 1994 inventory and the current inventory (column 9). For cases where an MPA was not available, the relative MPA was generated by multiplying the reported inventory by 10. If the relative MPA is larger than the previously calculated maximum activity, or smaller than the minimum activity, the relative MPA was then used to replace the maximum or minimum activity. This added conservatism was done to maintain some consistency between the current approach of using reported inventories vs the previous approach of using MPAs to model the source term. The mode (column 13) is the reported inventory value. Columns 14 and 15 contain the final minimum and maximum activity values. The nuclear data used to convert the MPAs from units of curies to grams are provided in Table A.28.

**Table A.28 Nuclear data for the non-IWMF
key radionuclides**

Nuclide	Half-life (days)	Specific activity (Ci/g)
^3H	4.50×10^3	9.67×10^3
^{14}C	2.09×10^6	4.46×10^0
^{90}Sr	1.04×10^4	1.39×10^2
^{99}Tc	7.78×10^7	1.69×10^{-2}
^{232}Th	5.13×10^{12}	1.10×10^{-7}
^{233}U	5.81×10^7	9.64×10^{-3}
^{235}U	2.57×10^{11}	2.16×10^{-6}
^{238}U	1.63×10^{12}	3.36×10^{-7}
^{237}Np	7.82×10^8	7.04×10^{-4}
^{239}Pu	8.81×10^6	6.20×10^{-2}
^{240}Pu	2.40×10^6	2.27×10^{-1}
^{241}Am	1.58×10^5	3.43×10^0
^{243}Am	2.70×10^6	1.99×10^{-1}

A.6 IWMF INVENTORY

A.6.1 Key Radionuclides

A review of the 63 radionuclides used to model the IWMF inventory showed that most radionuclide activities were well below the groundwater based activity limits, i.e., within 1% of the groundwater activity limit. Application of the Most Probable Activity Factor results in exceedance of the groundwater activity limit for ^{14}C and ^{239}Pu . Technitium-99, ^{233}U , and ^{241}Am were below the activity limits even when the Most Probable Activity Factors are used. However, the ratio of current inventory to limit for these radionuclides are sufficiently high that the sum of fractions (inventory to limit) for all the 63 radionuclides will exceed 1. These 5 radionuclide and 8 others, chosen based on sensitivity (in terms of analyses and significance in terms of dose contribution, were retained for uncertainty estimation. The 13 radionuclides are:

The activities of the remaining radionuclides modeled in this revised PA were very small (even when the upper bound of the activity uncertainties were considered) relative to the activity limits. For these radionuclides, the estimated uncertainties (where applicable) from the initial version of the PA are considered adequate in view of the relatively small amounts of activity involved.

A.6.2 Uncertainty in the IWMF Inventory

The source term uncertainties were investigated and updated (from the initial PA) for the 13 radionuclides identified above. When uncertainty estimation was being performed for the initial PA, the IWMF had only been in operation for over one-and-a-half years. Since then, the amount of activity placed on the IWMF has increased significantly in most cases. Thus, an update for the key radionuclides was deemed necessary to investigate the changes, if any, in waste characterization and reporting practices. Improvements in waste characterization will reduce the uncertainties associated with the reported waste inventories. The method used to update the waste inventory uncertainties for the 13 radionuclides is listed as follows:

1. Obtained the list of IWMF waste containers containing the radionuclides of interest .
2. Sorted the containers by activity for each radionuclide.
3. Identified the few containers containing the most activity for each radionuclide (i.e., the few containers which contain the 60 to 90% of IWMF activity inventory for a particular radionuclide).
4. Obtained a list of items in each of the containers identified in Step 3.

5. Reviewed the disposal records for each item. Identified the method used to determine the reported activity and determined the quality of the reported values.

The results of the uncertainty update are provided in Table A.29.

A trend that was generally observed is that the quality of reported radionuclide activities in the waste has improved since the initial PA. Sampling and analysis and assay measurements are being required for the radioactive waste, as well as better documentation of process knowledge. Notes from the review of the five most sensitive radionuclides (relative to the activity limits), out of the 13 key radionuclides, are provided below.

Table A.29 Summary of Updated Inventory Review Results for the IW MF

Nuclide	Percent of inventory reviewed	Basis of reported activity	Quality of reported activities
^3H	97% (1 container)	Analysis by SEG	Good
^{14}C	69% (7 containers)	Mostly 6CE, with 1 container analyzed by SEG (2.1 mCi, 9% of the inventory)	Poor
^{99}Tc	90% (3 containers)	6CE	Poor
$^{90}\text{Sr}^a$	n/a	n/a	n/a
^{232}Th	89% (1 container)	6CE and generator estimation	Poor
^{233}U	48% (3 containers)	Lab analysis	Good
^{235}U	90% (2 containers)	6CE and generator estimation	Poor
^{238}U	69% (1 container)	Calculation based on weight %	Average
^{237}Np	66% (1 container)	6CE	Poor
^{239}Pu	61% (3 containers)	Lab analysis	Good
^{240}Pu	98% (1 container)	Lab analysis	Good
^{241}Am	75% (7 containers)	Analyzed (49% of inventory), 6CE for the remaining inventory reviewed	Average
^{243}Am	88% (2 containers)	SEG analysis (60% of inventory), 6CE for the remaining inventory reviewed	Average

^a ^{90}Sr activity is spread over many containers.

²³⁹Pu

- Reviewed the data for 3 containers (X10C9401248, X10C9501864, X10C9502282) which make up 61% of the IWMF ²³⁹Pu inventory. There were 29 items in the 3 containers. There were 113 containers with ²³⁹Pu on the IWMF at the time the records were reviewed.
- The waste was mainly ion exchange resin (in plastic drums), contaminated debris wrapped in plastic, and pieces of process line piping. The chemical form was reported as oxide.
- Based on the records (Waste Item Description - UCN-2109) and a discussion with the generator, it was confirmed that all the items were sampled and lab analyzed. Pieces of piping were cut and sent to a laboratory for analysis.
- Uncertainty in reported laboratory result will arise from sampling uncertainty and analytical uncertainty.
- The Most Probable Activity Factor is 78. The original results were underestimated (the radionuclide has a low yield photon emission). The estimated inventory was derived by bounding the weight of the waste by 10 nCi/g. At the time of the initial PA, the IWMF ²³⁹Pu inventory was 5.98E-04 Ci. This represents only 4.6% of the current IWMF inventory.
- The Most Probable Activity Factor in this case is meaningless when applied to other waste packages since it is based solely on the mass of the waste.

²³⁸U

- Reviewed the data for 1 container (X10C9600218) which makes up 69% of the IWMF ²³⁸U inventory. There were 19 items in the container. There were 224 containers with ²³⁸U on the IWMF at the time the records were reviewed.
- Based on the records (Waste Item Description - UCN-2109), the waste appears to be mainly depleted uranium powder, U₃O₈.
- In item X10I9513969, there appears to be a revised gram quantity of ²³⁸U that was not reflected in the total U-238 mass, and hence activity, for the container. The correct ²³⁸U activity for the container should be 2.35E-02 Ci as opposed to the 3.04E-02 Ci reported.
- The Most Probable Activity Factor is 21.4. The original results were underestimated. There is relatively minimal beta/gamma emission by daughters. A modified dose rate conversion factor was used to estimate the most probable activity. At the time of the initial PA, the IWMF ²³⁸U inventory was 2.71E-03 Ci. This represents only 6.2% of the current reported IWMF inventory or 7.3% of the "corrected" inventory (where the mistake is accounted for).
- The records of one of the containers examined shows that the mass of the waste was used for determining the activity. The mass was converted to activity using a specific activity

constant for ^{238}U . Any errors in the activity reported would arise from an error in the mass value. The magnitude of this error can only be speculated. From the records, it appears that the weight was estimated at 100 kg and the reported number used was derived by subtracting oxygen from the U_3O_8 .

 ^{233}U

- Reviewed the data for 3 containers (X10C9402223, X10C9502282, X10C9402941) which at $4.0\text{E-}03$ Ci makes up 48% of the IWMF ^{233}U inventory. There were 125 items in the 3 containers. There were 128 containers with ^{233}U on the IWMF at the time the records were reviewed.
- The waste was mainly pieces of piping removed from Building 3019A. The chemical form was reported as oxide.
- Based on the records (Waste Item Description - UCN-2109) and a discussion with the generator, it was confirmed all the items were sampled and lab analyzed. Pieces of piping were cut and sent to lab for analysis.
- The Most Probable Activity Factor is 86.5. The original results were underestimated. There is relatively minimal beta/gamma emission by daughters. The estimated inventory was derived by bounding the weight of the waste by 10 nCi/g. At the time of the initial PA, the IWMF ^{233}U inventory was $1.07\text{E-}03$ Ci. This represents only 12.8% of the current reported IWMF inventory.
- The Most Probable Activity Factor in this case is meaningless when applied to other waste packages since it is based solely on the mass of the waste.

 ^{14}C

- Reviewed the data for the following 7 containers: X10CSLLN07521, X10CSLLN07525, X10CSLLN07526, X10CSLLN08523, X10CSLLN07522, X10CSLLN08535, X10C9400929. These containers contain 69% of the IWMF ^{14}C inventory. There were 51 containers with ^{14}C on the IWMF at the time the records were reviewed.
- The ^{14}C activity in the containers above, with the exception of container X10C9400929, were estimated using the 6CE method.
- Container X10C9400929 contains approximately 9% of the IWMF ^{14}C inventory. The reported ^{14}C activity in container X10C9400929 was based on analytical results provided by Scientific Ecology Group (SEG).
- The waste was mainly pieces of material and equipment (wood, plastic, paper, metal).
- The Most Probable Activity Factor is 500. This number was a pure estimate based on the potential magnitude of the error in the reported inventory, i.e., could be greater than 100, but maybe less than 1,000. Hence, 500. Note that the 500 was used as a measure of conservatism. Since the error was estimated to be between 2 to 3 orders of magnitude, 0.05

could have been used instead (though it would not be conservative). Another consideration is that ^{14}C is relatively difficult (i.e., nearly impossible) to measure with an "open-window" ion chamber. Thus, any relatively minor reading detected with an ion chamber corresponds to a very large amount of ^{14}C .

^{99}Tc

- Reviewed data for 3 containers (X10CSLLN007529 [3.4E-03 Ci reported], X10CSLLN008523 [3.4E-03 Ci reported], and X10C9400932 [4.07E-5 Ci]). The containers contain 90% of the IWMF ^{99}Tc inventory. There were 32 containers with ^{99}Tc on the IWMF at the time the records were reviewed.
- The waste was reported as metal, plastic, cloth, and rubber. The chemical form was reported as oxide.
- The rationale for reporting the presence of ^{99}Tc is not well founded. The method for reporting the quantity of ^{99}Tc was based on the 6CE rule, using an "unknown" instrument to perform the dose rate measurement. When combined, these "unknowns" lead to large uncertainties. The 6CE rule is not applicable to ^{99}Tc . And it is doubtful that the HP Tech used a proper instrument (that responds to the presence of ^{99}Tc) to conduct the dose rate measurement. Breaking this down by container:

X10CSLLN007529

Technitium-99 is reported in conjunction with 15 packets from the same building, 3029. Using an ion chamber, the dose rate was measured for each packet. Each packet had only 1 radionuclide, one of four: ^{137}Cs , ^{244}Cm , ^{99}Tc , or ^{192}Ir . There is likely to be large uncertainty as to the guess of the radionuclide present in each packet, since it is highly unlikely that the waste was segregated to that level (i.e. by radionuclide). Two packets (#9 and #10) had measured dose rates of 1 mrem/h, with the corresponding ^{99}Tc activity reported as 1.7E-04 Ci (Activity = $1/6 \times \text{Dose Rate \{mrem/h\}}/1000$). There are several questions that arise:

1. What process generated this odd "mix" of radionuclides?
2. How were the radionuclides isolated into individual packets?
3. Did the HP Tech really measure the bag to be 1 mrem/h, did he measure background, or were there other radionuclides in the bag? If the HP Tech measured 1 mrem/h in a bag full of pure ^{99}Tc , then the 6CE underestimates the real activity by over a factor of 1000. This argument is dismissed on the basis that measuring any ^{99}Tc (short of many thousands of curies) with a "closed-window" ion chamber is incredible. It is much more likely that either the background radiation rate was measured {since the majority of packets are marked as 1 mrem/h} or other unreported radionuclides in the single packet (perhaps ^{137}Cs) actually contributed to the measured dose rate. In

either case, there is no direct measurement that allows one to confirm or question the presence of ^{99}Tc in a single packet. There is a complete lack of knowledge on the basis for reporting the presence of ^{99}Tc and computing its activity.

X10CSLLN008523

The quantity of ^{99}Tc reported in this container was derived from the 6CE rule as well. The same conclusions apply to this container.

- Uncertainty in reported ^{99}Tc is from lack of knowledge. There is no measurable statistical error.
- The Most Probable Activity Factor is 432. The rationale used in the initial PA was to accept the generator's dose rate measurement. In the single reported case of ^{99}Tc , it was not found in a mixture with other radionuclides. A dose rate factor of 73.5 Ci/R/h was derived and applied to the measurement. When the dose-rate estimated activity content was divided by the reported inventory, the result was a MPAF of 432. At the time, this factor was thought to provide sufficient conservatism in the huge uncertainty in the data. As pointed out above, the dose to curie conversion for ^{99}Tc may lead to huge error much greater than 432. On the other hand, we feel that this type of error to be so erroneous that it is believed that the dose rate was caused by background (packets of waste in the surrounding area) or by some other radionuclide, not reported, in the packet.
- The Most Probable Activity Factor in this case is meaningless when applied to other waste packages since it is based solely on ratio of the reported ^{99}Tc activity to the "hypothetical" dose to curie conversion factor developed for one package.
- Technitium-99 is reported in a small fraction of IWMF containers. The bulk of the activity is reported in only a few containers. The basis for indicating the presence of ^{99}Tc and for determining the quantity is poorly understood. If the dose rate from ^{99}Tc was really measured, there are many curies in the facility (rather than a few millicuries). Fortunately, it seems incredible to expect ^{99}Tc in reasonable quantities to register a response on an ion chamber. The generators must have had a good reason for reporting the presence of ^{99}Tc , so its presence cannot be dismissed completely.

^{241}Am

- Reviewed the data for 7 containers (X10CSLLN008109, X10CSLLN008110, X10CSLLN008115, X10C9502282, X10CSLLN006642, X10C9501864, and X10CSLLN006643)) which make up 81% of the IWMF ^{241}Am inventory. There were 114 containers with ^{241}Am on the IWMF at the time the records were reviewed.
- The waste was mainly filters and contaminated laboratory debris.
- Based on the records (Waste Item Description - UCN-2109), it was observed that two of the containers, with approximately 40% of the total disposed ^{241}Am activity, were characterized

using the MSHLD code for dose rate verification. The two containers were generated in 1992. The remaining containers were characterized using the 6CE method and were generated in 1991.

- At least 40% of the ^{241}Am inventory appears to be well characterized.
- The Most Probable Activity Factor is 30. The original results were underestimated (the radionuclide has a low yield photon emission). The estimated inventory was derived by bounding the weight of the waste by 10 nCi/g. At the time of the initial PA, the IWMF ^{241}Am inventory was $2.21\text{E-}04$ Ci. This represents only 0.7% of the current IWMF inventory.
- The Most Probable Activity Factor in this case is meaningless when applied to other waste packages since it is based solely on the mass of the waste.

A.7 REFERENCES

Crystal Ball® for Windows™ 1993. Version 3.0, Decisioneering, Inc., Denver Colorado.

Iman, R. L., and M. J. Shortencarrier 1984. *A Fortran 77 Program and User's Guide for the Generation of Latin Hypercube and Random Samples for Use with Computer Models.*, NUREG/CR-3624, SAND83-2365, Sandia National Laboratory, Albuquerque, New Mexico.

ORNL (Oak Ridge National Laboratory) 1994. *Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6*, ORNL-6783, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

Shleien, B. 1989. *The Health Physics and Radionuclide Health Handbook*, Scinta, Inc., Silver Spring, Maryland.

Appendix B

DESCRIPTION OF THE SOURCE1 AND SOURCE2 COMPUTER PROGRAMS

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Appendix B

DESCRIPTION OF THE SOURCE1 AND SOURCE2 COMPUTER PROGRAMS

Version 2.0 of the SOURCE1 and SOURCE2 computer codes (Icenhour and Tharp 1996) were used to calculate source terms for this performance assessment. The original version of these computer codes, Version 1.0, was developed by Rogers and Associates Engineering Corporation (Shuman 1992). This appendix provides an overview of the SOURCE1 and SOURCE2 (collectively called SOURCE) computer codes. A more detailed description of the codes (including algorithms, input requirements, and output options) can be found in the user's manual (Icenhour and Tharp 1996).

The conceptual and mathematical modeling methodology used in the SOURCE computer codes is discussed in the following subsections. This discussion considers the approaches taken in (1) modeling concrete degradation, (2) performing the structural and cracking analyses for the disposal structures, (3) partitioning water through the disposal unit, and (4) modeling advective and diffusive releases of radionuclides from waste.

B.1 CONCRETE DEGRADATION MODELING

Modes of concrete degradation are considered in terms of surface- and bulk-attack mechanisms. Surface-attack mechanisms are initiated at the surface of the concrete component and progress inward, over time. Bulk-attack mechanisms modify the properties of the entire concrete component uniformly.

Sulfate attack is generally considered the most significant surface attack mechanism in the context of waste repositories (Atkinson and Hearne 1984). In areas characterized by cold winters, freeze-thaw cycling may also represent a serious threat to concrete in disposal facilities. In terms of the bulk-attack processes, the most notable degradation processes are likely to be calcium hydroxide leaching and alkali-aggregate attack.

Corrosion of reinforced steel may also undermine the ability of engineered disposal facilities to isolate waste therein from the environment. This process differs from the surface- and bulk-attack processes noted previously because it does not directly alter the properties of the concrete.

The models used in simulating concrete degradation are discussed in Sects. B.1.1 through B.1.3. The deterioration processes considered in the SOURCE computer codes include sulfate attack, calcium hydroxide leaching, and corrosion of steel reinforcement.

B.1.1 Sulfate Attack

Sulfate attack generally manifests itself in the form of expansion and, ultimately, cracking of concrete. It may also result in a progressive loss of strength and mass resulting from deterioration in the cohesiveness of the cement hydration products.

Three steps are recognized in the deterioration of concrete as a result of sulfate attack (Atkinson and Hearne 1990):

- Sulfate ions from the environment penetrate into the concrete, usually by diffusion.
- Sulfate ions react expansively with certain aluminum-containing phases in the concrete to form ettringite.
- The resulting internal expansion causes stress, cracking, and exfoliation of the concrete surface.

These aspects of the degradation process are incorporated into the sulfate attack model used in the SOURCE computer codes.

The sulfate attack model is based on the work of Atkinson and Hearne (1990). In this model, the reaction zone is assumed to spall out when it reaches a critical thickness given by

$$X_{\text{spall}} = \frac{2\alpha\gamma(1 - \mu_c)}{E(\beta C_e)^2}, \quad (\text{B.1})$$

where

- X_{spall} = reaction zone thickness at which spalling occurs (m),
- α = roughness factor for fracture path (unitless),
- γ = fracture surface energy of concrete (J/m²),
- μ_c = Poisson's ratio for concrete (unitless),
- E = Young's modulus (Pa),
- β = linear strain caused by a mole of sulfate reacted in 1 m³ (m³/mol), and
- C_e = concentration of sulfate as ettringite at the time at which spalling occurs (mol/m³).

This critical thickness is achieved at a time

$$t_{\text{spall}} = \frac{X_{\text{spall}}^2 C_e}{2D_i c_o}, \quad (\text{B.2})$$

where

- t_{spall} = time at which spalling occurs (s),

- D_i = "intrinsic" diffusion coefficient of sulfate ions in water-saturated cement (m^2/s),
and
 c_o = groundwater sulfate concentration (mol/m^3).

The rate of degradation, R , is defined as $X_{\text{spall}}/t_{\text{spall}}$ (m/s). Then, based on Eqs. (B.1) and (B.2), R is given by

$$R = \frac{E\beta^2 c_o C_e D_i}{\alpha\gamma(1 - \mu_c)} \quad (\text{B.3})$$

As sulfate attack progresses into the concrete member, it is assumed that the affected layers spall off, thus effectively reducing the thickness of the concrete member.

It is necessary to use an iterative method to determine the concentration of sulfate as ettringite, C_e , and the degradation rate caused by sulfate attack. The starting approximation for C_e is calculated assuming that the alumina has been completely converted to ettringite in the reacted zone (Atkinson and Hearne 1990). Zero-order values of the X_{spall} , t_{spall} , and R are calculated on this basis, and t_{spall} is compared with the time required for the reaction to go to completion. If t_{spall} is not great enough to permit complete reaction, t_{spall} and C_e are iterated to self-consistency using the reaction kinetics expression described in Atkinson and Hearne (1990).

B.1.2 Calcium Hydroxide Leaching

Calcium hydroxide [$\text{Ca}(\text{OH})_2$] leaching results in a loss of strength in the concrete as well as a lowering of the pH of the material. A loss of strength will affect the concrete structure's ability to withstand the loads placed upon it. Declines in the pH of the concrete may lead to depassivation of the steel reinforcement, thereby promoting corrosion of the steel.

$\text{Ca}(\text{OH})_2$ may be leached from the concrete because of diffusion and advection. The loss of $\text{Ca}(\text{OH})_2$ from concrete members from diffusion is calculated by solving the following equation:

$$\frac{df_t}{dt} = - D_y \frac{d^2 f_t}{dy^2} \quad (\text{B.4})$$

where

- f_t = fraction of $\text{Ca}(\text{OH})_2$ remaining in concrete member as a function of position and time (unitless),
 t = time (s),
 D_y = effective diffusion coefficient of $\text{Ca}(\text{OH})_2$ in concrete (m^2/s), and
 y = distance from centerline (m).

The initial conditions that apply are

$$f_i(y, t = 0) = \begin{cases} 1 & \text{for } |y| < \frac{\hat{W}}{2} \\ 0 & \text{for } |y| > \frac{\hat{W}}{2} \end{cases}, \quad (\text{B.5})$$

where

$\hat{W}$ = width of concrete member (m).

The equation for f_i is given by

$$f_i = 0.5 \operatorname{erf} \frac{(y + \hat{W}/2)R_f}{2(D_y t)^{0.5}} - 0.5 \operatorname{erf} \frac{(y - \hat{W}/2)R_f}{2(D_y t)^{0.5}}, \quad (\text{B.6})$$

where

R_f = retardation factor for $\text{Ca}(\text{OH})_2$ in concrete (unitless).

The $\text{Ca}(\text{OH})_2$ retardation factor, R_f , is given by

$$R_f = 1 + \frac{\text{Ca}_c}{\epsilon_c \text{Ca}_p}, \quad (\text{B.7})$$

where

Ca_c = $\text{Ca}(\text{OH})_2$ concentration in concrete (mol/L),

ϵ_c = concrete porosity (dimensionless), and

Ca_p = $\text{Ca}(\text{OH})_2$ concentration in concrete pore solution (mol/L).

The potential for leaching of $\text{Ca}(\text{OH})_2$ through advective mechanisms will depend upon the nature of the groundwater. If the groundwater is saturated or supersaturated with calcium carbonate, no dissolution of $\text{Ca}(\text{OH})_2$ will occur. Groundwater which is not saturated with calcium carbonate may leach $\text{Ca}(\text{OH})_2$ as the groundwater passes through the concrete.

Langelier (1936) has developed the calcium carbonate saturation (or Langelier) index as a means of characterizing the degree of calcium carbonate saturation of groundwater. The index takes into account the effects of temperature, total dissolved solids, total alkalinity, pH, and calcium content on the saturation characteristics of the groundwater. A negative value for the Langelier index denotes a groundwater which is not saturated with calcium carbonate [i.e., one capable of leaching $\text{Ca}(\text{OH})_2$ from concrete]. Index values equal to or greater than zero indicate calcium carbonate saturation.

Data taken from Langelier (1936) are used in a regression to estimate the saturation index as a function of the total dissolved solids and temperature of the groundwater. When predicted values of the index are positive, losses of Ca(OH)_2 are modeled to occur as a result of diffusion only. When the groundwater is not saturated with calcium carbonate (i.e., the Langelier index is negative), advective leaching of Ca(OH)_2 is calculated using (Atkinson 1985)

$$\text{Ca}_1 = I \frac{\text{Ca}_p}{C_t \text{Ca}_c}, \quad (\text{B.8})$$

where

- Ca_1 = fractional groundwater release rate of Ca(OH)_2 (year^{-1}),
- I = water percolation rate through vault (m/year), and
- C_t = concrete-member thickness (m).

The presence of other ions in the groundwater may influence the rate at which Ca(OH)_2 is leached from the concrete. Atkinson (1985) reports that magnesium and carbonate are among the species most likely to speed the loss of Ca(OH)_2 . The effect of these species is modeled using Eq. (B.8), replacing the pore solution concentration of Ca(OH)_2 , Ca_p , with the sum of this concentration and the groundwater concentrations of magnesium and carbonates.

The quantities of Ca(OH)_2 lost from the concrete member by diffusion and advection are summed to determine the total amount of the constituent leached from the concrete. The concentration of Ca(OH)_2 in the concrete is adjusted downward to reflect these losses. All Ca(OH)_2 leached from the concrete is assumed to be drawn from the calcium-silicate-hydrate (C-S-H) system of the concrete. The calcium incorporated into the relatively less-soluble phases of the concrete is not considered.

Changes in the pH of concrete as a result of the loss of Ca(OH)_2 have been well documented (Atkinson 1985). The pH of the concrete is maintained at levels greater than approximately 12.5 in the presence of alkalis, NaOH, and KOH. As these highly soluble species are lost because of leaching, the pH declines until it reaches 12.5, at which point the pH of the concrete is controlled primarily by the Ca(OH)_2 content of the concrete.

Changes in the pH of the concrete are modeled as alkalis and Ca(OH)_2 are leached from the concrete. Based on the data of Greenberg and Chang (1965), the pH is modeled to decline linearly from the initial pH of the concrete, as specified by the user, to 12.5 in direct proportion to the reduction in NaOH and KOH in the concrete. The rates of loss of these species from leaching by diffusion and advection are calculated using Eqs. (B.6) and (B.8), respectively.

Following the complete loss of NaOH and KOH from the concrete, changes in the pH of the concrete are modeled as a function of the Ca(OH)_2 content. Using data from the work of Greenberg and Chang (1965), concrete pH was regressed on the Ca:Si ratio of the material. As Ca(OH)_2 is leached from the concrete, the Ca:Si ratio is updated, and the pH of the concrete is estimated using this regression.

In addition to the pH effects noted, the loss of Ca(OH)_2 will result also in a reduction in the strength of the concrete. The loss in strength has been estimated to be approximately 1.5% for every 1.0% of the Ca(OH)_2 leached from the concrete (Lea 1970). Based on this relationship, the compressive strengths of the concrete members are updated to reflect losses of Ca(OH)_2 .

B.1.3 Corrosion of Steel Reinforcement

The damage to concrete resulting from the corrosion of steel reinforcement manifests itself in expansion, cracking, and spalling of the concrete member. The reinforced concrete member may suffer structural damage because of (a) the loss of the bond between the steel and concrete and (b) the loss of reinforcement cross-sectional area.

Steel reinforcement is generally passivated because of the alkaline nature of the liquid phase in the concrete pores and, hence, does not undergo corrosion. The passive layer on the steel may be destroyed through a direct lowering of the pH of the concrete via carbonation or because of chloride ion penetration to the steel. Both mechanisms of depassivation are considered in the SOURCE computer codes.

Carbonation of the concrete occurs as a result of the diffusion of carbon dioxide (CO_2) into the material. The depth of carbonation is given by (Tuutti 1982)

$$X = kt^{0.5} \quad , \quad (\text{B.9})$$

where

- X = depth of carbonation (m) and
- k = carbonation coefficient ($\text{m/s}^{0.5}$).

Given the value of k, the depth of penetration of the carbonation front into the concrete can be determined for any specified time.

The carbonation coefficient is calculated using (Tuutti 1982)

$$\frac{C_x - C_s}{g(k/2D_{\text{CO}_2}^{0.5})} + C_x - C_f = 0 \quad , \quad (\text{B.10})$$

where

- C_x = concentration of CO_2 bound in concrete (mol/L),
- C_s = CO_2 concentration at surface of concrete (mol/L),
- C_f = CO_2 concentration ahead of carbonation front (mol/L), and
- $g(k/2D_{\text{CO}_2}^{0.5})$ = function, where
 D_{CO_2} = diffusion coefficient of CO_2 in concrete (m^2/s).

The function $g(k/2D_{\text{CO}_2}^{0.5})$ is given by

$$g\left(\frac{k}{2D_{CO_2}^{0.5}}\right) = \pi^{0.5} \frac{k}{2D_{CO_2}^{0.5}} \exp\left(-\frac{k^2}{4D_{CO_2}}\right) \operatorname{erf}\left(\frac{k}{2D_{CO_2}^{0.5}}\right) . \quad (B.11)$$

Eqs. (B.10) and (B.11) are combined to arrive at a solution for k , the carbonation coefficient. Assuming that the concentration of CO_2 ahead of the carbonation front is zero (i.e., that the carbonation front is discontinuous), this solution can be simplified to yield

$$k^2 = \frac{4D_{CO_2}}{\pi^{0.5}} \left(\frac{C_s}{C_x} - 1 \right) . \quad (B.12)$$

When using Eq. (B.12) to evaluate k , the model takes into account that a portion of the CO_2 diffusing into the concrete is bound by concrete constituents and does not penetrate to the steel reinforcement. This bound CO_2 plays no role in depassivation. The amount of CO_2 bound in the concrete is set equal to amount of hydrated lime in the concrete (Tuutti 1982). The quantity of hydrated lime is calculated as the product of the CaO content in the concrete and the degree of hydration. The degree of hydration, estimated based on the water/cement ratio for Portland cements (Tuutti 1982), is given by

$$H_f = 0.4 + 0.5(WCR) , \quad (B.13)$$

where

H_f = fraction of hydrated CaO (unitless) and
 WCR = water-cement ratio (unitless).

Given the carbonation coefficient, k , the depth of carbonation is calculated using Eq. (B.9). As stated previously, this solution assumes that the carbonation front is discontinuous. At that time when the front has penetrated to the depth of the steel reinforcement, depassivation of the steel is assumed to occur, and corrosion is thus initiated.

Steel reinforcement may also be depassivated as a result of the penetration of chloride ions to the steel surface. Using a standard solution to Fick's first law of diffusion, the chloride ion concentration at the steel is calculated as

$$CL_s = CL_i + (CL_{gw} - CL_i) \left[1 - \operatorname{erf}\left(\frac{C_c}{2(D_{Cl}t)^{0.5}}\right) \right] , \quad (B.14)$$

where

- CL_s = chloride ion concentration at steel reinforcement (mol/L),
- CL_i = initial chloride ion concentration in concrete (mol/L),
- CL_{gw} = chloride ion concentration in groundwater (mol/L),
- C_c = concrete cover thickness (m), and
- D_{Cl} = effective diffusivity of chloride in concrete (m^2/s).

The concentration of chloride ions at the steel reinforcement required to depassivate the steel has been considered by numerous investigators. Hausmann (1967) found that the pH of the concrete had an effect on the level of chloride ions required to initiate corrosion. In studies carried out using NaOH and $Ca(OH)_2$ solutions, it was found that a chloride ion to hydroxide ion concentration ratio of 0.61 was sufficient to depassivate the steel.

Using the results of Hausmann, a chloride-ion-to-hydroxide-ion ratio of 0.61 is assumed to result in depassivation of the steel. The hydroxide ion concentration used in calculating this ratio is determined using the modeled concrete pH. As discussed in Sect. B.1.2, the pH of the concrete changes with time as NaOH, KOH, and $Ca(OH)_2$ are leached from the material.

Upon depassivation of the steel reinforcement by either carbonation or chloride penetration, corrosion is modeled to occur at a rate determined by the rate of diffusion of oxygen to the steel. The molar flow of oxygen to the surface of the steel reinforcement is modeled using Fick's first law of diffusion:

$$J_o = - D_o A \frac{d[O_2]}{dx}, \quad (B.15)$$

where

- J_o = oxygen flux at the steel reinforcement (g/s),
- D_o = effective diffusivity of oxygen through concrete (cm^2/s),
- A = surface area over which oxygen diffuses to the reinforcement (cm^2), and
- $\frac{d[O_2]}{dx}$ = dissolved oxygen concentration gradient (g/cm^4).

The rate of oxygen consumption by the corrosion reaction is assumed to be greater than the rate of oxygen diffusion to the reaction surface. Under these conditions, the corrosion rate is limited by the flux of oxygen at the steel reinforcement.

The use of epoxy-coated steel reinforcement may delay the onset of corrosion by isolating the steel from aggressive ions and oxygen. The use of epoxy coating is not assumed to delay the time of depassivation of the reinforcement. However, upon depassivation, the coating is assumed to prevent corrosion as long as it remains intact.

The effectiveness of epoxy coating on steel reinforcement is modeled using a linear failure function. Using the time at which failure of the coating begins and the time required for all epoxy coating to fail, a fraction of the reinforcement coating which has failed is calculated. This failure fraction is used to linearly adjust the projected rate of corrosion downward.

Corrosion of components other than steel reinforcement will also affect the long-term performance of LLW disposal facilities. For example, the metal boxes placed inside tumulus vaults, the corrugated steel liners used in the construction of disposal silos, and the cast iron pipes used in well construction will all eventually fail as a result of corrosion.

Corrosion of steel and iron barriers used in the tumulus, silo, well-in-silo, and well disposal technologies is considered in the SOURCE computer codes. Failure rates of these barriers are modeled using linear failure functions. The user specifies the time at which corrosion of the metal component begins and the number of years required, following this time, for the member to fail completely. Using these input data, a failure fraction is calculated for each year of the simulation.

B.2 CONCRETE STRUCTURAL AND CRACKING ANALYSES

The structural and cracking analyses serve two distinct purposes in modeling the long-term performance of LLW disposal units. The structural analysis considers the loads placed on the disposal unit to determine the bending moments, shears, axial tension, and compressive forces placed on the various structural components. Because these loads vary with the structural component under consideration, this analysis is carried out for the roof, walls, and floor of each disposal unit.

The cracking analyses are concerned with the ability of the disposal unit to bear the loads placed upon it. Bending moments, shears, axial tensions, and compressive forces calculated as part of the structural analysis are compared with loads and forces at which structural failure will occur to determine the structural integrity of the disposal unit. The cracking analyses must account for the changes in concrete properties projected to occur because of physical and chemical attack. As such, it is conducted for each year of the simulation, or until hydraulic failure of the disposal unit is complete.

The structural and cracking analyses must address the structural features of each disposal unit. Even though the disposal silos and wells share a number of common features, both of these technologies differ significantly from the tumulus technology. Consequently, the following discussion considers the silos and wells, and tumulus separately.

The models used in the structural and cracking analyses are generally applicable to performance analysis of concrete, steel, and iron structural components. However, the manner in which some of these models are applied is specific to the disposal technologies in use at SWSA 6. Consequently, general application of the SOURCE computer codes to other disposal configurations requires extreme caution.

B.2.1 Tumulus Technology

The long-term performance of a tumulus disposal unit is a function of the performance of the individual vaults of which it is composed. The structural and cracking analyses performed to model the behavior of these vaults are discussed in Sects. B.2.1.1 and B.2.1.2, respectively. Additionally, models for the degradation and failure of the tumulus pad and leachate collection system are presented in Sects. B.2.1.3 and B.2.1.4.

B.2.1.1 Structural Analysis

Each structural component of the vaults has unique loading conditions placed upon it. As such, separate structural analyses are conducted for the roof or lid, walls, and floor of the vaults. A description of these analyses follows.

B.2.1.1.1 Vault roof

The roof is analyzed structurally as a simply supported, or hinged, slab. The uniform load on the roof of a vault in layer i of the tumulus disposal unit is calculated as

$$q_r = s\rho_s + ih_r\rho_c + (i-1)[h_w\rho_w + (h_f + h_r)\rho_c] , \quad (\text{B.16})$$

where

q_r	=	uniform load on vault roof in layer i (lb/in. ²),
s	=	soil cover thickness (in.),
ρ_s	=	density of soil cover (lb/in. ³),
i	=	layer number (unitless),
h_r	=	roof thickness (in.),
ρ_c	=	density of reinforced concrete (lb/in. ³),
h_w	=	waste thickness (in.),
ρ_w	=	density of waste (lb/in. ³), and
h_f	=	floor thickness (in.).

Thermal loads on the vaults are not considered because the insulating properties of the cover material minimize thermal gradients across the concrete structural components.

The deflection of the simply supported rectangular roof resulting from the uniform load is given by

$$W(x, y) = \frac{16q_r w_r}{\pi^6 D_r} \sum_{m=1}^{\infty} \sum_{n=1}^{\infty} \frac{\sin \frac{m\pi x}{a} \sin \frac{n\pi y}{b}}{mn \left(\frac{m^2}{a^2} + \frac{n^2}{b^2} \right)^2}, \quad (B.17)$$

where

- $W(x, y)$ = deflection of roof at location (x, y) (in.),
 w_r = width of roof in (x, y) direction (in.),
 m, n = 1, 3, 5, ...,
 a = width of waste cell (in.),
 b = length of waste cell (in.), and
 D_r = flexural rigidity of the roof (lb-in.²).

The flexural rigidity of the roof is calculated using

$$D_r = \frac{E_c h_r^3 w_r}{12(1 - \mu_c^2)}, \quad (B.18)$$

where E_c = modulus of elasticity of concrete (lb/in.²).

Based on Eqs. (B.17) and (B.18), the bending moments resulting from uniform loading as a function of location on the roof are calculated as

$$M_x = q_r a^2 \left[\frac{16}{\pi^4} \sum_{m=1}^{\infty} \sum_{n=1}^{\infty} \frac{\sin \frac{m\pi x}{a} \sin \frac{n\pi y}{b}}{\frac{n}{m} \left(m^2 + \frac{n^2}{\left(\frac{b}{a}\right)^2} \right)^2} + \mu_c \frac{\sin \frac{m\pi x}{a} \sin \frac{n\pi y}{b}}{\left(\frac{b}{a}\right)^2 \left(\frac{m}{n}\right) \left(m^2 + \frac{n^2}{\left(\frac{b}{a}\right)^2} \right)^2} \right] \quad (B.19)$$

and

$$M_y = q_r b^2 \left[\frac{16}{\pi^4} \sum_{m=1}^{\infty} \sum_{n=1}^{\infty} \frac{\sin \frac{m\pi x}{a} \sin \frac{n\pi y}{b}}{\frac{m}{n} \left(\left(\frac{a}{b} \right)^2 + n^2 \right)^2} \right. \\ \left. + \mu_c \frac{\sin \frac{m\pi x}{a} \sin \frac{n\pi y}{b}}{\left(\frac{a}{b} \right)^2 \left(\frac{n}{m} \right) \left(\left(\frac{a}{b} \right)^2 + n^2 \right)^2} \right], \quad (B.20)$$

where

- M_x = bending moment resulting from uniform loading in the x-direction parallel to width of roof (lb-in./in.) and
 M_y = bending moment resulting from uniform loading in the y-direction parallel to length of roof (lb-in./in.).

Uniform loads on the vault roof result in shear forces upon that component as well. These forces are calculated as

$$Q_x = q_r a \left\{ \frac{16}{\pi^3} \sum_{m=1}^{\infty} \sum_{n=1}^{\infty} \cos \frac{m\pi x}{a} \cdot \sin \frac{n\pi y}{b} \right. \\ \left. \left[\frac{m^2}{n \left(m^2 + n^2 \left(\frac{a}{b} \right)^2 \right)^2} + \frac{n}{\left(m^2 \left(\frac{b}{a} \right)^2 + n^2 \left(\frac{a}{b} \right)^2 \right)^2} \right] \right\} \quad (B.21)$$

and

$$Q_y = q_r b \left\{ \frac{16}{\pi^3} \sum_{m=1}^{\infty} \sum_{n=1}^{\infty} \sin \frac{m\pi x}{a} \cos \frac{n\pi y}{b} \left[\frac{m}{\left(m^2 \left(\frac{b}{a} \right)^2 + n^2 \left(\frac{a}{b} \right)^2 \right)^2} + \frac{n^2}{m \left(m^2 \left(\frac{b}{a} \right)^2 + n^2 \right)^2} \right] \right\} , \quad (B.22)$$

where

- Q_x = shear force resulting from uniform loading in the x-direction (lb/in.) and
 Q_y = shear force resulting from uniform loading in the y-direction (lb/in.).

B.2.1.1.2 Vault walls

The vault walls are subject to vertical, or uniform, loads and hydrostatic pressures. The uniform loads are calculated for a vault in layer i of the tumulus disposal unit using

$$q_w = \rho_s \left(\frac{h_r}{2} i + (h_f + h_w)(i - 1) + s \right) (1 - \sin f_s) , \quad (B.23)$$

where

- q_w = uniform load on vault wall in layer i (lb/in.²) and
 f_s = friction angle of soil backfill around tumulus (deg).

Hydrostatic pressures on the vault walls are a result of lateral soil pressures from the soil backfill and from the waste and grout inside the vault. This pressure is calculated as

$$P = \rho_s \left(h_w + \frac{h_r + h_f}{2} \right) (1 - \sin f_s) - \rho_w \left(h_w + \frac{h_r + h_f}{2} \right) (1 - \sin f_w) , \quad (B.24)$$

where

- P = maximum hydrostatic pressure (lb/in.²) and
 f_w = friction angle of waste (deg).

Bending moment calculations for the vault walls must account for the uniform loads and hydrostatic pressures on the structural components. Bending moments resulting from the uniform load

are calculated using Eqs. (B.19) and (B.20), substituting the uniform load on the wall for the uniform load on the roof and changing roof dimensions to those of the wall. Bending moments caused by hydrostatic pressures are calculated using

$$M_{xh} = Pa^2 \sum_{m=1}^{\infty} (m\pi)^2 \left[\frac{2(-1)^{m+1}}{\pi^5 m^5} + [(1-\mu_c)A_m - 2\mu_c B_m] \cosh\left(\frac{m\pi y}{a}\right) + (1-\mu_c)B_m \left(\frac{m\pi}{a}\right) y \sinh\left(\frac{m\pi y}{a}\right) \right] \sin \frac{m\pi x}{a} \quad (B.25)$$

and

$$M_{yh} = Pa^2 \sum_{m=1}^{\infty} (m\pi)^2 \left[\frac{2(-1)^{m+1}}{\pi^5 m^5} \mu_c + [(\mu_c-1)A_m - 2B_m] \cosh\left(\frac{m\pi y}{a}\right) + (\mu_c-1)B_m \left(\frac{m\pi}{a}\right) y \sinh\left(\frac{m\pi y}{a}\right) \right] \sin \frac{m\pi x}{a} \quad , \quad (B.26)$$

where

- M_{xh} = bending moment resulting from hydrostatic pressures in the x-direction parallel to the width of the wall (lb-in./in.) and
 M_{yh} = bending moment resulting from hydrostatic pressures in the y-direction parallel to the length of the wall (lb-in./in.).

The quantities A_m and B_m are given, respectively, by

$$A_m = - \frac{(2 + \alpha_m \tanh \alpha_m)(-1)^{m+1}}{\pi^5 m^5 \cosh \alpha_m} \quad (B.27)$$

and

$$B_m = \frac{(-1)^{m+1}}{\pi^5 m^5 \cosh \alpha_m} \quad , \quad (B.28)$$

where

$$\alpha_m = \frac{m\pi b}{2a} \quad (B.29)$$

Bending moments calculated for the uniform load and hydrostatic pressure on the wall are summed to arrive at the final bending moments for the wall as a function of location. The calculations of the bending moments are repeated for each wall geometry comprising the disposal vault.

Shear forces caused by hydrostatic pressures on the vault walls are calculated as

$$Q_{xh} = 2Pa \sum_{m=1}^{\infty} (m\pi)^3 \left[\frac{(-1)^{m+1}}{\pi^5 m^5} - B_m \cosh\left(\frac{m\pi y}{a}\right) \right] \cos\left(\frac{m\pi x}{a}\right) \quad (B.30)$$

and

$$Q_{yh} = -2Pa \sum_{m=1}^{\infty} (m\pi)^3 B_m \sinh\left(\frac{m\pi y}{a}\right) \sin\left(\frac{m\pi x}{a}\right) \quad (B.31)$$

where

$$\begin{aligned} Q_{xh} &= \text{shear force resulting from hydrostatic loading in the x-direction (lb in) and} \\ Q_{yh} &= \text{shear force resulting from hydrostatic loading in the y-direction (lb in)} \end{aligned}$$

The shear forces on vault walls resulting from uniform loads are calculated using Eqs. (B.21) and (B.22) and substituting appropriate parameters for the wall. Shear forces caused by uniform and hydrostatic loads are summed for each wall location. Calculations of shear forces are performed for each wall geometry comprising the vault.

The walls of the vaults are subject to compressive forces from the roof reaction and the weight of the walls. The compressive force, calculated as a function of height on the wall, is given by

$$F_w = R_{ry} + h_{wl} z \rho_c \quad (B.32)$$

where

$$\begin{aligned} F_w &= \text{compressive force on wall at height } z \text{ (lb/in.)}, \\ R_{ry} &= \text{roof reaction in y direction at height } z \text{ (lb/in.)}, \\ h_{wl} &= \text{thickness of wall (in.)}, \text{ and} \\ z &= \text{wall height (in.)} \end{aligned}$$

B.2.1.1.3 Vault floor

The floor of a given vault must bear loads from the walls, including the wall weight and loads transmitted to the walls from the roof, and loads from the waste within the vault. Based on the floor geometry illustrated in Fig. B.1, the bending moments in the region x (or y) $\leq a$ of the beam subjected to a concentrated force and moment are calculated as

$$M_x = \frac{P_x I_{1px}}{2\lambda_x [\sinh^2(\lambda_x w_f) - \sin^2(\lambda_x w_f)]} \quad (\text{B.33})$$

and

$$M_y = \frac{P_y I_{1py}}{2\lambda_y [\sinh^2(\lambda_y l_f) - \sin^2(\lambda_y l_f)]}, \quad (\text{B.34})$$

where

- M_x = bending moment in x-direction parallel to width of floor (lb-in./in.),
- P_x = applied concentrated load caused by wall in x-direction (lb/in.),
- I_{1px} = trigonometric function (unitless),
- $\lambda_x = \left[\frac{\tilde{k} l_f}{4D_{fx}} \right]^{0.25} (\text{in.}^{-1})$,
- $\tilde{k}$ = modulus of the subgrade reaction (lb/in.³),
- l_f = length of floor (in.),
- D_{fx} = flexural rigidity of floor in the x-direction (lb-in.²),
- w_f = width of floor (in.),
- M_y = bending moment in y-direction parallel to length of floor (lb-in./in.),
- P_y = applied concentrated load caused by wall in y-direction (lb/in.),
- I_{1py} = trigonometric function (unitless),
- $\lambda_y = \left[\frac{\tilde{k} w_f}{4D_{fy}} \right]^{0.25} (\text{in.}^{-1})$, and
- D_{fy} = flexural rigidity of floor in the y-direction (lb-in.²).

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Fig. B.1 Schematic diagram of floor geometry used in the calculation of bending moments.

Applied moments caused by the wall are not considered in the bending moment calculations because it is assumed that the floor and walls are hinged.

The concentrated load on the floor, P_x or P_y , is calculated as a function of location using

$$P_x = R_{rx} + h_{wl} \left(h_w + \frac{h_r + h_f}{2} \right) (\rho_c - \rho_w) \quad (\text{B.35})$$

and

$$P_y = R_{ry} + h_w \left(h_w + \frac{h_r + h_f}{2} \right) (\rho_c - \rho_w) , \quad (B.36)$$

where

R_{rx} = roof reaction in x-direction (lb/in.).

The flexural rigidity of the floor is calculated using Eq. (B.18). To calculate the quantity D_x the thickness and width of the floor are substituted for h_r and w_r respectively. The thickness and length of the floor are substituted for h_f and w_f respectively, to calculate D_y .

The parameters I_{ipx} and I_{ipy} are complex trigonometric functions of λ_x , λ_y , and the geometry of the structural member and are given by

$$\begin{aligned} I_{ipx} = & 2 \sinh(\lambda_x x) \sin(\lambda_x x) \left[\sinh(\lambda_x w_f) \cos(\lambda_x a) \cosh(\lambda_x c) \right. \\ & - \sin(\lambda_x w_f) \cosh(\lambda_x a) \cos(\lambda_x c) \left. \right] - \left[\sinh(\lambda_x x) \cos(\lambda_x x) \right. \\ & - \cosh(\lambda_x x) \sin(\lambda_x x) \left. \right] \left[\sinh(\lambda_x w_f) \{ \sin(\lambda_x a) \cosh(\lambda_x c) \right. \\ & - \cos(\lambda_x a) \sinh(\lambda_x c) \} + \sin(\lambda_x w_f) \{ \sinh(\lambda_x a) \cos(\lambda_x c) \\ & - \cosh(\lambda_x a) \sin(\lambda_x c) \} \left. \right] \end{aligned} \quad (B.37)$$

and

$$\begin{aligned} I_{ipy} = & 2 \sinh(\lambda_y y) \sin(\lambda_y y) \left[\sinh(\lambda_y l_f) \cos(\lambda_y a) \cosh(\lambda_y c) \right. \\ & - \sin(\lambda_y l_f) \cosh(\lambda_y a) \cos(\lambda_y c) \left. \right] - \left[\sinh(\lambda_y y) \cos(\lambda_y y) \right. \\ & - \cosh(\lambda_y y) \sin(\lambda_y y) \left. \right] \left[\sinh(\lambda_y l_f) \{ \sin(\lambda_y a) \cosh(\lambda_y c) \right. \\ & - \cos(\lambda_y a) \sinh(\lambda_y c) \} + \sin(\lambda_y l_f) \{ \sinh(\lambda_y a) \cos(\lambda_y c) \\ & - \cosh(\lambda_y a) \sin(\lambda_y c) \} \left. \right] , \end{aligned} \quad (B.38)$$

where the parameters a and c are indicated in Fig. B.1.

Shear forces on the floor of the vault are calculated as a function of location using

$$Q_x = \frac{P_x I_{2px}}{\left[\sinh^2(\lambda_x w_f) - \sin^2(\lambda_x w_f) \right]} \quad (B.39)$$

and

$$Q_y = \frac{P_y I_{2py}}{\left[\sinh^2(\lambda_y l_f) - \sin^2(\lambda_y l_f) \right]}, \quad (B.40)$$

where

$$\begin{aligned} I_{2px} &= \text{trigonometric function (unitless) and} \\ I_{2py} &= \text{trigonometric function (unitless).} \end{aligned}$$

The parameters I_{2px} and I_{2py} are functions of λ_x , λ_y , and the geometry of the structural member and are given by

$$\begin{aligned} I_{2px} = & \left[\cosh(\lambda_x x) \sin(\lambda_x x) + \sinh(\lambda_x x) \cos(\lambda_x x) \right] \left[\sinh(\lambda_x w_f) \right. \\ & \left. \cos(\lambda_x a) \cosh(\lambda_x c) \sin(\lambda_x w_f) \cosh(\lambda_x a) \cos(\lambda_x c) \right] \\ & + \sinh(\lambda_x x) \sin(\lambda_x s) \left[\sinh(\lambda_x w_f) \{ \sin(\lambda_x a) \cosh(\lambda_x c) \right. \\ & \left. - \cos(\lambda_x a) \sinh(\lambda_x c) \} + \sin(\lambda_x w_f) \{ \sinh(\lambda_x a) \cos(\lambda_x c) \right. \\ & \left. - \cosh(\lambda_x a) \sin(\lambda_x c) \} \right] \end{aligned} \quad (B.41)$$

and

$$\begin{aligned}
I_{2py} = & [\cosh(\lambda_y y) \sin(\lambda_y y) + \sinh(\lambda_y y) \cos(\lambda_y y)] [\sinh(\lambda_y w_f) \\
& \cos(\lambda_y a) \cosh(\lambda_y c) \sin(\lambda_y l_f) \cosh(\lambda_y a) \cos(\lambda_y c)] \\
& + \sinh(\lambda_y y) \sin(\lambda_y s) [\sinh(\lambda_y l_f) \{\sin(\lambda_y a) \cosh(\lambda_y c) \\
& - \cos(\lambda_y a) \sinh(\lambda_y c)\} + \sin(\lambda_y l_f) \{\sinh(\lambda_y a) \cos(\lambda_y c) \\
& - \cosh(\lambda_y a) \sin(\lambda_y c)\}]
\end{aligned} \tag{B.42}$$

B.2.1.2 Cracking Analyses

The cracking analyses are performed to assess the ability of the structural components of each vault to bear the loads placed upon it. In the event that the roof, wall(s), or floor of a vault cannot bear these loads, cracking will occur. Cracking of these components may occur as a result of shear forces or bending; cracking of the vault walls may also result from compressive loads on the structure. The manner in which these modes of cracking are modeled is discussed in the following.

Shear cracking will occur if the shear force on a concrete member exceeds the cracking shear of the member. The cracking shears for the roof, floor, and wall in the horizontal direction are calculated as the minimum of

$$V_{cr} = d_t (1.9 C_{str}^{0.5} + 2500 S_{tn} Q_x / M_x) \tag{B.43}$$

or

$$V_{cr} = 3.5 C_{str}^{0.5} d_t, \tag{B.44}$$

where

- V_{cr} = shear force at which cracking occurs (lb/in.),
- d_t = distance from steel reinforcement in tension to compression face of concrete (in.),
- C_{str} = compressive strength of concrete (lb/in.²), and
- S_{tn} = area of steel reinforcement in tension per unit width (in.²/in.).

The cracking shear for the wall in the vertical direction is the minimum of Eqs. (B.43) and (B.45).

$$V_{cr} = 3.5 C_{str}^{0.5} d_t (1. + F_w / (500 h_{wl}))^{0.5} . \quad (B.45)$$

In the event of cracking caused by shear failure, crack characteristics are determined. The depth of the single crack is the thickness of the concrete member, and the crack width is 0.013 in.

Cracking caused by bending will be initiated if the bending moments calculated for a given concrete member exceed the cracking moment for that structural component. The cracking moment is given by

$$M_{cr} = \frac{I_g f_r}{y_t a_u} , \quad (B.46)$$

where

$$\begin{aligned} M_{cr} &= \text{cracking moment per unit width} \left(\frac{\text{lb-in.}}{\text{in.}} \right), \\ I_g &= \text{moment of inertia of concrete section (in.}^4\text{)}, \\ f_r &= \text{modulus of rupture (lb/in.}^2\text{)}, \\ y_t &= \text{distance from the centroidal axis to the tensile face of the concrete (in.), and} \\ a_u &= \text{unit width of concrete member.} \end{aligned}$$

For a rectangular slab

$$I_g = \frac{a_u h_m^3}{12} , \quad (B.47)$$

where

$$h_m = \text{concrete member thickness (in.).}$$

The value of y_t is given by

$$y_t = \frac{h_m}{2} . \quad (B.48)$$

Axial compression force is conservatively neglected in the roof and floor.

If the bending moments exceed the cracking moment but do not exceed the ultimate strength of the concrete member, cracks will not propagate through the entire member. If, however, the bending moments exceed the ultimate strength of the structural component, cracks will span the thickness of the member. The ultimate flexural strength of a member without compressive steel is approximated using

$$M_u = \phi S_{tn} f_y \left(d_t - \frac{C_d}{2} \right) \quad , \quad (B.49)$$

where

- M_u = ultimate flexural strength (lb-in./in.),
- ϕ = strength reduction factor (unitless),
- f_y = yield strength of steel reinforcement (lb/in.²), and
- C_d = depth of the compression block (in.).

The depth of the compression block is calculated using

$$C_d = \frac{S_{tn} f_y}{0.85 C_{str}} \quad . \quad (B.50)$$

If all reinforcement has been lost because of corrosion, the ultimate strength is equal to the cracking moment.

Crack characteristics are calculated as fractures due to loading and propagate through a given structural component. The depth of cracking caused by bending is calculated as the distance from the surface of the concrete to the neutral axis. Crack depth is computed using the strain compatibility relation wherein the tensile crack depth is given by

$$d_{cr} = \frac{d_t}{\frac{St_n}{E_s} + \frac{St_m}{E_c}} \left[\frac{St_n}{E_s} + \epsilon_{sh} \right] + d_c \quad , \quad (B.51)$$

where

- d_{cr} = crack depth (in.),
- ϵ_{sh} = shrinkage strain of concrete (in./in.),
- d_c = concrete cover thickness on tension face (in.),
- St_n = tensile stress in steel reinforcement (lb/in.²),
- St_m = maximum concrete compressive stress (lb/in.²), and
- E_s = modulus of elasticity of steel reinforcement (for this application, taken as 200,000 MPa) (MPa).

The tensile stress in steel reinforcement is calculated using

$$St_n = \left(\frac{E_s}{E_c} \right) M \left(\frac{d_t - R_t}{I_e} \right) , \quad (B.52)$$

where

M = bending moment due to uniform loading in x or y direction (lb-in./in.) and
 I_e = effective moment of inertia per unit width of concrete member (in.³).

The quantity R_t (in.) is given by

$$R_t = \frac{- (\hat{\beta}_1 + \hat{\beta}_2) + [(\hat{\beta}_1 + \hat{\beta}_2)^2 - 4\alpha (C_c \hat{\beta}_1 - d_t \hat{\beta}_2)]^{0.5}}{2\alpha_1} . \quad (B.53)$$

The quantities α_1 (unitless), $\hat{\beta}_1$ (in.), and $\hat{\beta}_2$ (in.) are calculated by

$$\alpha_1 = 0.5 , \quad (B.54)$$

$$\hat{\beta}_1 = \left(\frac{\pi}{S_{p2}} \right) \left(\frac{S_{d1}}{2} \right)^2 \left(\frac{E_s}{E_c} - 1 \right) , \quad (B.55)$$

and

$$\hat{\beta}_2 = \left(\frac{\pi}{S_{p2}} \right) \left(\frac{S_{d1}}{2} \right)^2 \left(\frac{E_s}{E_c} \right) , \quad (B.56)$$

where

S_{d1} = diameter of steel reinforcement in direction 1, and closest to concrete outer tension face (in.) and
 S_{p2} = spacing of steel reinforcement in direction 2, perpendicular to direction 1 (in.).

The quantity I_e is given by

$$I_e = \left(\frac{1}{a_u} \right) \left[\left(\frac{M_{cr}}{M} \right)^3 I_g + \left(1 - \left(\frac{M_{cr}}{M} \right)^3 \right) I_c \right], \quad (B.57)$$

where

I_c = cracking moment of inertia in the x- or y-direction (in.⁴).

The cracking moment of inertia is calculated using the following equation:

$$I_c = a_u \left[0.333 R_t^3 + \hat{\beta}_1 (R_t - C_c)^2 + \hat{\beta}_2 (d_t - R_t)^2 \right]. \quad (B.58)$$

The maximum concrete compressive stress is given by

$$St_m = M \left(\frac{R_t}{I_c} \right). \quad (B.59)$$

In modeling the water flow characteristics of failed concrete, it is assumed that cracks achieving a depth equal to three-fourths of the remaining slab thickness functionally penetrate the slab. Prior to this, flow through the concrete is the same as that through intact concrete. If the bending moment exceeds the ultimate strength of the concrete slab, cracks penetrate immediately through the slab.

Numerous equations have been proposed for the prediction of crack spacing and width in flexural members. Nawy (1966) developed a formula for calculating mean crack spacing for a two-way concrete slab:

$$S_{m1} = 0.5 K_n \left(\frac{S_{dt} S_{p2}}{Q_d} \right)^{0.5}, \quad (B.60)$$

where

S_{m1} = mean crack spacing in direction 1 (in.).

The variables K_n and Q_d are given by

$$K_n = \left[1.6 + 2.4 \left(\frac{a}{b} - 0.5 \right) \right] 0.29 \quad (\text{B.61})$$

and

$$Q_d = \frac{S_{tn}}{24 d_c} \quad (\text{B.62})$$

If the bending moment exceeds the cracking moment, but not the ultimate strength of the concrete member, the mean crack width is given by

$$W_m = S_m \left(\beta_r \frac{S_{tn}}{E_s} + \epsilon_{sh} \right) \quad (\text{B.63})$$

where

W_m = mean crack width (in.) and

S_m = mean crack spacing (in.).

The quantity β_r is given by

$$\beta_r = \frac{d_{cr}}{d_{cr} - d_c} \quad (\text{B.64})$$

If the bending moment exceeds the ultimate strength of the concrete member, the crack width is calculated as

$$W_m = S_m \left(\frac{f_y}{E_s} + \epsilon_{sh} \right) \quad (\text{B.65})$$

If the compressive forces on the wall exceed the ultimate strength of the wall in compression, cracking will occur. The ultimate strength of the wall in compression is calculated as

$$M_{uc} = 0.39 h_{wl} C_{str} \left[1 - \left(\frac{h_c}{32 h_{wl}} \right)^2 \right] , \quad (B.66)$$

where

M_{uc} = ultimate strength of the wall in compression (lb/in.) and
 h_c = height of vault wall (in.).

Cracking because of compression results in a single crack extending through the concrete member; the crack width is one-tenth of the height of the wall section under consideration.

Cracking of a reinforced concrete member may occur also as a result of corrosion of the steel reinforcement. As the concrete surrounding the reinforcement prevents free expansion, the steel-corrosion products will exert pressure within the concrete. Based on elasticity theory (Saada 1974), the magnitude of this internal pressure is approximated using

$$P_i = \frac{\Delta}{r_o} \frac{1}{\left[\frac{1-\mu_r}{E_r} + \frac{(1-\mu_c)r_o^2 + (1+\mu_c)d_{cv}^2}{E_c(d_{cv} - r_o^2)} \right]} , \quad (B.67)$$

where

P_i = internal pressure due to corrosion (lb/in.²),
 Δ = thickness of the free expansion layer (in.),
 r_o = original radius of steel reinforcement (in.),
 μ_r = Poisson's ratio of corrosion product (unitless),
 d_{cv} = distance from concrete face to center of steel reinforcement (in.), and
 E_r = modulus of elasticity of corrosion product (lb/in.²).

The thickness of the free expansion layer is given by

$$\Delta = r_c + C_{tc} - r_o , \quad (B.68)$$

where

- r_c = radius of remaining steel reinforcement (in.) and
 C_{tc} = thickness of corrosion layer under conditions of free expansion (in.).

A general series form of the stress function in bipolar coordinates was given by Jeffrey (1920). This function has been applied to the situation of a semi-infinite region with a circular hole under a uniform radius pressure (Fig. B.2). Based on this work, the stress on the surface of the concrete is given by

$$\sigma_x = -4P_i \frac{r_o^2 (x^2 - d_{cv}^2 + r_o^2)}{(x^2 + d_{cv}^2 - r_o^2)^2}, \quad (B.69)$$

where

- σ_x = stress at surface of concrete (lb/in.²) and
 x = distance from point A (Fig. B.2) along the surface of the concrete (in.).

The point of maximum stress occurs at point A (Fig. B.2), where $x = 0$. For this case, Eq. (B.69) reduces to

$$\sigma_{xA} = \frac{4P_i r_o^2}{(d_{cv}^2 - r_o^2)}. \quad (B.70)$$

The tangent stress at any point, Q or Q', around the circular hole is given by

$$\sigma_{\theta Q} = P_i (1 + 2 \tan^2 \phi), \quad (B.71)$$

where

- $\sigma_{\theta Q}$ = tangent stress at point Q (lb/in.²).

Fig. B.2 Schematic diagram of steel and concrete geometry used in calculating stress resulting from corrosion of steel reinforcement.

This relationship exhibits a maximum tangent stress at point E and is calculated using

$$\sigma_{\theta E} = P_i \frac{d_{cv}^2 + r_o^2}{d_{cv}^2 - r_o^2}, \quad (B.72)$$

where $\sigma_{\theta E}$ = maximum tangent stress (lb/in.²).

The magnitude of the maximum stress around the circular hole is a function of the ratio of the concrete cover thickness and the radius of the remaining steel reinforcement. The following dependencies are noted:

$$\begin{aligned} \text{if } d_{cv} < 1.73r, \quad \sigma_{xA} > \sigma_{\theta E} > 2P_i; \\ \text{if } d_{cv} = 1.73r, \quad \sigma_{xA} = \sigma_{\theta E} = 2P_i; \\ \text{if } d_{cv} > 1.73r, \quad \sigma_{xA} < \sigma_{\theta E} < 2P_i. \end{aligned}$$

These relationships provide a simple method for determining where cracking from corrosion will begin.

When the concrete cover thickness is greater than three times the diameter of the reinforcing steel, the tensile stresses around the circular boundary will approach the applied pressure P_i . Plain concrete has minimal tensile strength to resist these stresses—only 6 to 8% of the specified compressive strength of concrete. Consequently, the maximum tension stress can readily exceed the tensile strength of concrete, at which point cracking begins.

Cracking from corrosion is typically initiated internally, along the circular boundary, as the ratio of concrete cover thickness to the original radius of the steel is usually greater than 1.73 (Jeffrey 1920). As corrosion progresses, accompanied by the deterioration of the concrete cover, cracking will propagate toward the surface of the concrete slab. When the tension stress at the concrete surface equals or exceeds the tensile strength of the concrete, the cracking will penetrate the concrete cover. This cracking will occur along the length of the steel reinforcement.

Spalling out of the concrete will occur if the concrete cover over the steel reinforcement is small ($d_{cv} \leq 1.73r_o$). Under these conditions, the stresses at the concrete surface exceed both the stresses at the steel surface and the tensile strength of concrete and spalling along the reinforcement occurs.

B.2.1.3 Concrete Pad Degradation Model

Tumulus-type disposal facilities may use a steel-reinforced concrete pad upon which disposal vaults are placed. This pad, while intact, should divert water to the leachate collection system. To incorporate the performance of the concrete pad into SOURCE1, a compressive failure

model is assumed. Failure is estimated by calculating the reinforcement ratio (Cowan 1982), which is defined by

$$\rho = \left(\frac{A}{b} \right) \frac{1}{d} , \quad (\text{B.73})$$

where ρ = reinforcement ratio (dimensionless),

$\left(\frac{A}{b} \right)$ = cross-sectional area of steel reinforcement per unit width of slab (m), and

d = effective depth of steel (distance from the top of the slab to the center of the steel reinforcement) (m).

The reinforcement ratio at which compressive failure may occur is called the limiting reinforcement ratio and is given by (Cowan 1982)

$$\rho_{\text{lim}} = \frac{\epsilon'_c}{\epsilon'_c + \epsilon_y} 0.85\beta_1 \frac{f'_c}{f_y} , \quad (\text{B.74})$$

where

ρ_{lim} = limiting reinforcement ratio (dimensionless),

ϵ'_c = ultimate concrete strain (for this application, taken as 0.003) (dimensionless),

ϵ_y = yield strain of steel (dimensionless),

β_1 = a factor used in the equivalent rectangular stress diagram for concrete at the ultimate load (dimensionless),

f'_c = specified compressive strength of concrete (MPa), and

f_y = specified yield strength of steel reinforcement (MPa).

The yield strain of the steel reinforcement can be calculated by

$$\epsilon_y = \frac{f_y}{E_s} , \quad (\text{B.75})$$

where E_s = modulus of elasticity of steel reinforcement (for this application, taken as 200,000 MPa) (MPa).

The value of β_1 is determined as follows (Cowan 1982):

$$\beta_1 = 0.85 \text{ for } f'_c \leq 30 \text{ MPa} \quad (\text{B.76})$$

or

$$\beta_1 = 0.85 - 0.08 \left(\frac{f'_c - 30}{10} \right) \text{ for } f'_c > 30 \text{ MPa} \quad (\text{B.77})$$

The values of the reinforcement ratio and the limiting reinforcement ratio are evaluated at annual time steps in SOURCE1. These two values are compared, and when the reinforcement ratio exceeds the limiting value, the pad is said to have failed hydraulically. Pad failure allows leachate to pass through it. Values of both ρ and ρ_{lim} will change because of the degradation of the concrete. The concrete is simulated to degrade by using the sulfate attack and calcium hydroxide leaching subroutines in SOURCE1. Sulfate attack results in the spalling off of the concrete cover on the reinforcing steel. Hence, as the effective depth of the steel decreases, the reinforcement ratio increases. Leaching of calcium hydroxide from the concrete pad results in reduced concrete strength. Therefore, as the compressive strength of the concrete decreases, the limiting reinforcement ratio decreases. Both of the concrete degradation mechanisms result in a decrease of the margin between the reinforcement ratio and the limiting reinforcement ratio, ultimately resulting in pad failure.

B.2.1.4 Leachate Collection System Degradation Model

Water that reaches an intact concrete pad of a tumulus-type disposal unit will be diverted to a leachate collection system. This system consists of piping, valves, collection sumps, and monitoring equipment. Ideally, with a properly functioning system, all leachate will be collected, and no release of radionuclides to the environment will occur.

The leachate collection system degradation model describes the functionality fraction of the collection system as a function of time. The functionality fraction is defined as the ratio of the amount of radionuclide in the collected leachate to the total radionuclide release from the disposal vaults and can vary from 0 to 1. With a value of 1, the leachate collection system is fully functional, and no radionuclides are released to the environment. A zero value indicates a fully degraded system that allows all leached radionuclides to be released to the environment.

The initial functionality fraction and the length of the institutional control period are input parameters to the SOURCE1 code. The functionality fraction degrades linearly to zero from the beginning of the simulation until the end of the institutional control period. The degradation of the collection system is assumed to result from piping and valve leaks or failures, flow obstructions within the system, leakage or overflow of collection sumps, degraded monitoring equipment, etc. At

the end of the institutional control period, no maintenance of the collection system is assumed to occur. Hence, no credit is taken for the collection system after the end of institutional control. Additionally, if the concrete pad is predicted to fail hydraulically before the end of institutional control, the functionality fraction is set to zero at the time of pad failure.

B.2.2 Disposal Silo and Well Technology

The structural and cracking analyses of the disposal silos, wells, and wells-in-silos are discussed in Sects. B.2.2.1 through B.2.2.2. Even though these analyses are similar in many respects to the analyses conducted for the tumulus disposal technology, features unique to these disposal units require additional modeling considerations.

B.2.2.1 Structural Analysis

The structural analysis of the silo and well disposal technologies considers the three structural components of each—loosely referred to as the roof, wall, and floor. These analyses differ depending upon whether the silo or well configuration is being considered. For the analysis of wells-in-silos, structural analyses of both the silo and well are performed.

B.2.2.1.1 Silo or well roof

A polar coordinate system is used in the structural analysis of the roof of the silo or well disposal unit. The roof is modeled as a simply supported circular plate under uniform loading. The load upon the roof is calculated as

$$q_r = h_r \rho_c + s p_s \quad , \quad (B.78)$$

where

$$q_r = \text{uniform load on silo or well roof (lb/in.}^2\text{)}.$$

The final bending moments on the roof caused by the uniform load consist of radial and tangential components, which are given by

$$M_r = \frac{q_r}{16} (3 + \mu_c) (r_s^2 - r^2) \quad (B.79)$$

where

and M_r = radial component of bending moment (lb-in./in.),

$$M_t = \frac{q_r}{16} \left[r_s^2 (3 + \mu_c) - r^2 (1 + 3\mu_c) \right] , \quad (\text{B.80})$$

M_t = tangential component of bending moment (lb-in./in.),

r_s = radius of silo or well (in.), and

r = distance from center of silo or well roof (in.).

These components are summed to arrive at the total bending moment at location (θ, r) , namely

$$M_x = M_r \cos^2 \theta + M_t \sin^2 \theta \quad (\text{B.81})$$

and

$$M_y = M_r \sin^2 \theta + M_t \cos^2 \theta . \quad (\text{B.82})$$

The shear force on the roof at distance d_t from the interior face of the wall is calculated using

$$Q = \frac{q_r \left(r_s - \frac{h_{wl}}{2} - d_t \right)}{2} , \quad (\text{B.83})$$

where

Q = shear force on roof of well or silo (lb/in.).

B.2.2.1.2 Silo or well wall.

The wall of the silo or well is subject to uniform and hydrostatic pressures. Setting the origin of the coordinate system at the mid-height of the wall, the uniform pressure or load is calculated as

$$q_w = \rho_s \left(s + \frac{h_r}{2} + \frac{h_s}{2} \right) (1 - \sin f_s) - \frac{h_s}{2} \rho_w (1 - \sin f_w) , \quad (\text{B.84})$$

where

- q_w = uniform load on silo or well wall (lb/in.²),
- h_s = silo or well height (in.),
- f_s = friction angle of soil backfill around silo or well (deg), and
- f_w = friction angle of waste inside silo or well (deg).

The corresponding maximum antisymmetrical hydrostatic pressure on the wall is calculated using

$$P = \frac{h_s}{2} (\rho_s (1 - \sin f_s) - \rho_w (1 - \sin f_w)) \quad (B.85)$$

Bending moments and shear forces due to uniform pressure are calculated for the silo or well wall using

$$M_y = \frac{q_w h_s^2}{4\alpha^2} \left(\frac{\sin\alpha \sinh\alpha}{\cos 2\alpha + \cosh 2\alpha} \cos\beta y \cosh\beta y - \frac{\cos\alpha \cosh\alpha}{\cos 2\alpha + \cosh 2\alpha} \sin\beta y \sinh\beta y \right) \quad (B.86)$$

and

$$Q_y = \frac{q_w h_s}{2\alpha} \left[\frac{\sin\alpha \sinh\alpha}{\cos 2\alpha + \cosh 2\alpha} (\cos\beta y \sinh\beta y - \sin\beta y \cosh\beta y) - \frac{\cos\alpha \cosh\alpha}{\cos 2\alpha + \cosh 2\alpha} (\cos\beta y \sinh\beta y + \sin\beta y \cosh\beta y) \right] \quad (B.87)$$

where

- M_y = bending moment resulting from uniform loading in the y-direction (lb-in./in.) and
- Q_y = shear force resulting from uniform loading in the y-direction (lb/in.).

The quantities α and β are given by

$$\alpha = \frac{\beta h_s}{2} \quad (\text{B.88})$$

and

$$\beta = \left[\frac{3(1-\mu_c^2)}{r_s^2 h_{wl}^2} \right]^{0.25} \quad (\text{B.89})$$

The silo and well walls are also subject to axial and ring compression forces. The axial compressive force on the silo wall is calculated using Eq. (B.32). This same equation is used for the well-wall calculation by substituting the density of the cast iron for the density of concrete. The ring compression force resulting from a uniform load is calculated as

$$N_\theta = q_w r_s \left[1 - \frac{2 \sin \alpha \sinh \alpha}{\cos 2\alpha + \cosh 2\alpha} \sin \beta y \sinh \beta y - \frac{2 \cos \alpha \cosh \alpha}{\cos 2\alpha + \cosh 2\alpha} \cos \beta y \cosh \beta y \right], \quad (\text{B.90})$$

where

$$N_\theta = \text{ring compression force (lb/in.}^2\text{)}.$$

The bending moments and shear forces resulting from antisymmetrical hydrostatic pressure are calculated using

$$M_y = \frac{Ph_s^2}{4\alpha^2} \left(\frac{\sin \alpha \cosh \alpha}{\cosh 2\alpha - \cos 2\alpha} \cos \beta y \sinh \beta y - \frac{\cos \alpha \sinh \alpha}{\cosh 2\alpha - \cos 2\alpha} \sin \beta y \cosh \beta y \right) \quad (\text{B.91})$$

and

$$Q_y = \frac{Ph_s}{2\alpha} \left[\frac{\sin\alpha \cosh\alpha}{\cosh 2\alpha - \cos 2\alpha} (\cos\beta y \cosh\beta y - \sin\beta y \sinh\beta y) - \frac{\cos\alpha \sinh\alpha}{\cosh 2\alpha - \cos 2\alpha} (\cos\beta y \cosh\beta y + \sin\beta y \sinh\beta y) \right] \quad (B.92)$$

The ring compression force caused by hydrostatic pressure is calculated using

$$N_\theta = -2Pr_s \left[\frac{y}{h_s} - \frac{\sin\alpha \cosh\alpha}{\cosh 2\alpha - \cos 2\alpha} \sin\beta y \cosh\beta y - \frac{\cos\alpha \sinh\alpha}{\cosh 2\alpha - \cos 2\alpha} \cos\beta y \sinh\beta y \right] \quad (B.93)$$

Bending moments, shear forces, and ring compression forces calculated for the uniform and anti-symmetrical hydrostatic pressures are summed at each location on the wall to arrive at the final values.

B.2.2.1.3 Silo or well floor

The circular floor plate is subjected to a distributed line load, or concentrated force, along its perimeter. This concentrated force is calculated as

$$F_c = R_r + h_{wt} h_s (\rho_c - \rho_w) \quad (B.94)$$

where

- F_c = concentrated force (lb/in.) and
- R_r = roof reaction (lb/in.).

The radial and tangential components of the bending moments for the circular floor are given by

$$M_r = - \frac{D}{L_{DK}^2} \left[C_1 Z_{2r} - C_2 Z_{1r} - \frac{L_{DK}}{r} (1 - \mu_c) (C_1 Z'_{1r} + C_2 Z'_{2r}) \right] \quad (B.95)$$

and

$$M_t = - \frac{D}{L_{DK}^2} \left[\mu_c (C_1 Z_{2r} - C_2 Z_{1r}) + \frac{L_{DK}}{r} (1 - \mu_c) (C_1 Z'_{1r} + C_2 Z'_{2r}) \right] \quad (B.96)$$

The final bending moments are calculated using Eqs. (B.81) and (B.82).

The maximum shear force on the floor is calculated using

$$Q_{\max} = - \frac{D}{L_{DK}^3} [C_1 Z'_{2r} - C_2 Z'_{1r}] \quad (B.97)$$

where

$Q_{\max}$ = maximum shear force on floor (lb/in.).

The quantities D , C_1 , C_2 , and L_{DK} are calculated, respectively, as

$$D = \frac{E_c h_f^3}{12(1 - \mu_c^2)} \quad (B.98)$$

$$C_1 = - \frac{F_c}{\tilde{k} L_{DK} \Psi} \left[Z_{1r} + \frac{L_{DK}}{r_s} (1 - \mu_c) Z'_{2r} \right] \quad (B.99)$$

$$C_2 = - \frac{F_c}{\tilde{k} L_{DK} \psi} \left[Z_{2r} - \frac{L_{DK}}{r_s} (1 - \mu_c) Z'_{1r} \right] , \quad (B.100)$$

and

$$L_{DK} = \left(\frac{D}{\tilde{k}} \right)^{0.25} . \quad (B.101)$$

The quantities Z_{1r} , Z_{2r} , Z'_{1r} , and Z'_{2r} are calculated using the following equations.

$$Z_{1r} = 1 + \sum_{n=1}^{\infty} (-1)^n \frac{\left(\frac{r/L_{DK}}{2} \right)^{4n}}{((2n)!)^2} , \quad (B.102)$$

$$Z_{2r} = \sum_{n=1}^{\infty} (-1)^n \frac{\left(\frac{r/L_{DK}}{2} \right)^{4n-2}}{((2n-1)!)^2} , \quad (B.103)$$

$$Z'_{1r} = \sum_{n=1}^{\infty} (-1)^n \frac{2n \left(\frac{r/L_{DK}}{2} \right)^{4n-1}}{((2n)!)^2} , \quad (B.104)$$

and

$$Z'_{2r} = \sum_{n=1}^{\infty} (-1)^n \frac{(2n-1) \left(\frac{r/L_{DK}}{2} \right)^{4n-3}}{((2n-1)!)^2} , \quad (B.105)$$

where

$$n = 1, 2, 3 \dots$$

The variable ψ , used in Eqs. (B.99) and (B.100), is calculated as

$$\psi = Z_{1r}Z'_{2r} - Z'_{1r}Z_{2r} + \frac{L_{DK}}{r_s}(1 - \mu_c)(Z_{1r}^{1/2} + Z_{2r}^{1/2}) \quad (B.106)$$

The quantities Z_{1r} , Z_{2r} , Z'_{1r} , and Z'_{2r} are calculated using Eqs. (B.102) through (3.105), substituting the silo or well radius, r_s , for the parameter r .

B.2.2.2 Cracking Analyses

Cracking or failure of the disposal silos, wells, and wells-in-silos occurs at the point at which the structural components can no longer bear the loads placed upon them. Cracking of the roof, wall, and floor of the silo or well may occur as a result of shear forces or bending. Cracking of the wall may occur as a result of compressive forces on the structure.

The cracking analyses for the disposal silos are similar to that performed for tumulus-type facilities in that these analyses model the initiation and propagation of cracks in concrete barriers and calculate fracture characteristics. In contrast, the cracking analyses for the wells simply determine when the roof, wall, or floor undergoes initial failure and does not calculate fracture characteristics.

Shear cracking of a silo or well occurs if the shear force on the structural member exceeds the cracking shear of the member. The cracking shears for the roof and floor in the silo and well are calculated using Eqs. (B.43) and (B.44).

The cracking shear for the silo wall in the vertical direction is the minimum of

$$V_{cr} = h_{wl} \left(1.9C_{str}^{0.5} + 2500S_{in}Q_x/M_{my} \right) \quad (B.107)$$

and

$$V_{cr} = 3.5C_{str}^{0.5} h_{wl} \left(1 + F_w/(500h_{wl}) \right)^{0.5}, \quad (B.108)$$

where

$$M_{my} = \text{modified moment (lb-in.)}$$

The modified moment, M_{my} , is given by

$$M_{my} = M_y - 0.38F_w h_{wl} \quad (B.109)$$

F_w is determined from Eq. (B.32).

A single fracture will extend through the entire concrete member because of shear cracking. The width of the fracture is 0.013 in. The shear force at which failure of the cast iron wall of the well occurs is given as

$$V_f = 0.7h_{wl}f_{ws} \quad (B.110)$$

where

$$\begin{aligned} V_f &= \text{shear force at which well wall fails (lb/in.) and} \\ f_{ws} &= \text{yield strength of cast iron (lb/in.}^2\text{).} \end{aligned}$$

The roof and floor of the silos and wells crack if the bending moment at a given location exceeds the cracking moment. The cracking moment is calculated using Eq. (B.46). Cracks will not extend through the entire member unless the bending moments exceed the ultimate strength of the member. The ultimate flexural strength of the roof and floor is calculated using Eq. (B.49). The ultimate strength for the silo wall is calculated using

$$\begin{aligned} M_u = \phi \left[S_{tn} f_y \left(d_t - \frac{C_d}{2} \right) + L_{tn} f_y \left(h_{wl} + \frac{L_{tn}}{2} - \frac{C_d}{2} \right) \right. \\ \left. + L_{cm} f_y \left(\frac{C_d}{2} + \frac{L_{cm}}{2} \right) \right] \quad (B.111) \end{aligned}$$

where

$$\begin{aligned} L_{tn} &= \text{thickness of corrugated steel liner on tension face (in.) and} \\ L_{cm} &= \text{thickness of corrugated steel liner on compression face (in.).} \end{aligned}$$

The depth of the compression block is given by

$$C_d = \frac{f_y (S_{tn} + L_{tn} - L_{cm})}{.85 C_{str}} \quad (B.112)$$

Fracture depth, spacing, and width are calculated as cracks initiate and propagate in concrete members comprising the silos and wells-in-silos. These characteristics are calculated using the approach discussed for the tumulus-type unit (Sect. B.2.1.2).

The wall of the silo or well may fail from axial or ring compression. In terms of the former, the silo wall will crack if the axial compression force on the member exceeds the ultimate strength of the wall in compression or critical buckling strength. The strength of the wall in axial compression is calculated as the minimum of

$$N_{ac} = 0.39h_{wl}C_{str} \quad (B.113)$$

and

$$N_{ac} = \frac{D_w}{w_f h_s^2} m^2 \pi^2 + E_c h_{wl} \frac{h_s^2}{r_s^2} m^2 \pi^2, \quad (B.114)$$

where

- N_{ac} = ultimate strength or critical buckling strength under axial compression (lb/in.),
- D_w = flexural rigidity of wall (lb/in.²), and
- m = 1, 2, 3... .

The flexural rigidity of the wall is calculated using Eq. (B.18), substituting the thickness and unit width of the wall for h_r and w_r , respectively.

If the ring compression force on the silo exceeds the ultimate or buckling strength of the wall, cracking will occur. The strength of the wall subject to ring compression is given by the minimum value calculated using

$$N_{rc} = \frac{D_w}{w_f r_s^2} \left(n^2 - 1 + \frac{2n^2 - 1 - \mu_c}{1 + A} \right) + \frac{E h_{wl}}{(n^2 - 1)(1 + A)^2}, \quad (B.115)$$

where

- N_{rc} = ultimate strength or critical buckling strength under ring compression (lb/in.) and
- n = 2, 3, 4... .

The parameter A is calculated using

$$A = \left(\frac{nh_s}{\pi r_s} \right)^2 \quad . \quad (B.116)$$

Compressive forces and bending moments on the wall of the disposal well may result in failure of the well. If the combined stresses on the wall exceed the yield strength of the cast iron, failure will occur. The combined stresses are calculated as

$$N_{ac} = \frac{F_w}{h_{wl}} + \frac{6M_y}{h_{wl}^2} \quad . \quad (B.117)$$

The wall will also fail if the ring compression force on the well wall exceeds the buckling strength of the wall. The ultimate strength of the wall subject to ring compression is the minimum of

$$N_{rc} = 2h_{wl} \left(2 \times 10^6 \frac{h_{wl}}{r_s} \right) \left| \left(1 - 33.3 \frac{h_{wl}}{r_s} \right) \right| \quad (B.118)$$

and

$$N_{rc} = 3 \times 10^4 h_{wl} \quad . \quad (B.119)$$

Cracking of reinforced concrete resulting from the corrosion of the steel reinforcement is modeled using the same methodology developed for the tumulus disposal unit. This portion of the cracking analyses is performed for the roof and floor of the disposal silo only. The walls of the silo and the roof and floor of the well do not contain steel reinforcement.

B.3 FLOW PARTITIONING

A benefit of the concrete engineered barriers used in the tumulus, silo, and well disposal units is the material's low hydraulic conductivity. When intact, the concrete largely prevents water

from contacting the disposed waste. As the concrete members deteriorate with time, cracks form and greater amounts of water may contact the waste. Eventually, the conductivity of the concrete will be no better than that of the soil backfill around the disposal unit.

To calculate radionuclide releases as a result of advection, it is necessary to estimate the amount of water percolating through the waste. The water entering a disposal area is divided into two components: a component which flows around the disposal unit and a component which contacts the waste.

The flow partitioning scheme used in the SOURCE computer codes is based on the assumption of a saturated steady-state system under a unit hydraulic gradient. Under these conditions, the amount of water percolating through the intact vaults, silos, and wells is equal to the saturated hydraulic conductivity of the concrete. As the concrete members deteriorate and crack, preferential flow of water through the fractures occurs at much greater rates.

Preliminary analyses conducted with the SOURCE computer codes have indicated that much of the ability of a disposal unit to exclude water is lost when fractures penetrate through one or more concrete members. Based on these observations, the amount of water percolating through the waste is set equal to the amount of water entering the disposal area when fractures first penetrate the disposal unit. From this point on, the amount of water contacting the waste is solely a function of the hydraulic characteristics of the site soils and soil backfill.

B.4 RADIONUCLIDE RELEASE MODELING

The SOURCE codes incorporate two mass-transport mechanisms (advection and diffusion) that are modeled in one dimension. The concentration that is calculated to be released by these two mechanisms cannot exceed the solubility limit of the assumed chemical form of a nuclide. Rates of release from disposal units which have not undergone significant structural failure will generally be low—that is, below detection limits. These releases are dependent largely on the relative water saturation of the waste and concrete and, for the most part, are the result of diffusion. As a disposal unit deteriorates and undergoes cracking, water may percolate more easily through the waste. Under these conditions, leaching of radionuclides by advection can accelerate and may overshadow leaching by diffusion.

Leaching of radionuclides by advection is directly proportional to the amount of water contacting the waste and inversely proportional to the degree to which radionuclides are sorbed by the waste matrix. An analytical expression in which the radionuclide inventory is updated at preset time-steps is used to evaluate advective leaching. Leaching by diffusion is calculated using the FLOTHRU computer program (a subroutine in the SOURCE codes) (ORNL 1994). A description of these two leaching mechanisms is provided in the following sections.

B.4.1 Advective Transport Model

The analytical model for advective transport is based on work presented in Baes and Sharp (1983). A detailed derivation of the model can be found in Icenhour (1995).

The total radionuclide release during a time-step is calculated by the following formula:

$$L = \frac{\lambda_L}{\lambda_L + \lambda_d} Q_0 \left[e^{-(\lambda_L + \lambda_d)t_1} - e^{-(\lambda_L + \lambda_d)t_2} \right] , \quad (\text{B.120})$$

where

- L = mass of radionuclide leached because of advection (g),
- λ_L = leach rate constant (s^{-1}),
- λ_d = radioactive decay constant (s^{-1}),
- Q_0 = initial mass of radionuclide in the waste (g), and
- t_1, t_2 = the bounds of the time period of interest (s).

The leach constant, λ_L , is given by

$$\lambda_L = \frac{q}{W\theta R_d} , \quad (\text{B.121})$$

where

- q = water infiltration rate (cm/s),
- W = waste thickness (cm),
- θ = relative saturation (i.e., volume of water in waste/volume of waste) (dimensionless), and
- R_d = retardation factor (dimensionless).

Finally, the retardation factor, R_d , can be calculated by the following equation:

$$R_d = 1 + \frac{\rho_b}{\theta} K_d , \quad (\text{B.122})$$

where

- ρ_b = bulk density of waste (g/cm^3) and
- K_d = distribution coefficient (mL/g).

B.4.2 Diffusion Transport Model

The diffusive transport model that is outlined in this section is a summary of the work by Nestor presented in ORNL (1994).

Consider the two-layer slab representation of a waste disposal unit presented in Fig. B.3. The inner layer, which is of half-thickness a , initially contains a contaminant with concentration C_0 . The outer layer, which has thickness $b-a$, is initially uncontaminated. This situation is analogous to the grouted waste initially placed inside an uncontaminated concrete vault (e.g., tumulus-type disposal). If there is little bulk fluid flow through this system, then diffusion will be the dominant transport mechanism. Diffusion equations can then be written for the inner and outer layers of the disposal unit. The concentration of contaminant in the inner layer is denoted by C_1 , while that in the outer layer is denoted by C_2 .

Fig. B.3 Representation of the system modeled by the FLOTHRU computer code.

The diffusion equation for the inner layer is

$$\frac{\partial C_1}{\partial t} = D_1 \frac{\partial^2 C_1}{\partial x^2} - \lambda_d C_1 \quad , \quad (B.123)$$

where

- C_1 = concentration of contaminant in the inner layer (g/cm³),
- D_1 = effective diffusion coefficient for the contaminant in layer 1 (cm²/s), and
- x = spatial position (cm).

Similarly, a diffusion equation can be written for the outer layer:

$$\frac{\partial C_2}{\partial t} = D_2 \frac{\partial^2 C_2}{\partial x^2} - \lambda_d C_2 \quad , \quad (B.124)$$

where

- C_2 = concentration of contaminant in the outer layer (g/cm³) and
- D_2 = effective diffusion coefficient for the contaminant in layer 2 (cm²/s).

Eqs. (B.123) and (B.124) can be solved with appropriate initial and boundary conditions. In this case, the initial conditions are

$$C_1(x, 0) = C_0 \text{ for } 0 \leq x < a \quad (B.125)$$

and

$$C_2(x, 0) = 0 \text{ for } a \leq x < b \quad (B.126)$$

The boundary conditions are

$$\left. \frac{\partial C_1}{\partial x} \right|_{x=0} = 0 \quad , \quad (B.127)$$

$$C_1(a,t) = C_2(a,t) \quad , \quad (B.129)$$

$$C_2(b,t) = 0 \quad , \quad (B.128)$$

and

$$D_1 \frac{\partial C_1}{\partial x} \bigg|_{x=a} = D_2 \frac{\partial C_2}{\partial x} \bigg|_{x=a} \quad . \quad (B.130)$$

The solution to Eqs. (B.123) through (B.130) is implemented using the FLOTHRU computer code (ORNL 1994), which is incorporated as a subroutine into the SOURCE codes.

B.4.3 Calculation of Total Radionuclide Release

In order to calculate the total amount of radionuclide leaching from a disposal unit using the SOURCE codes, the advective and diffusive components are determined separately. These two components are then added together to calculate the total release. This calculated total is compared with the solubility limit of the radionuclide for the amount of water flowing through the unit. If this limit is exceeded, then the release is limited to the amount determined by solubility.

Radionuclides leached from the waste will be transported away from the disposal unit with the water percolating through the disposal unit. Two flow components are observed on the Oak Ridge Reservation (ORR). A vertical component represents recharge to the underlying aquifer at the site, while a lateral subsurface component discharges to surface waters.

Radionuclide releases from the disposal units are partitioned between the recharge and lateral flow components in proportion to the vertical and lateral fluxes. The amount of water which flows vertically to the aquifer is calculated as the minimum of the amount of water percolating through the disposal unit and the saturated hydraulic conductivity of the site soils. Water in excess of the saturated hydraulic conductivity is modeled as lateral sub-surface flow.

Based on the assumption that radionuclide concentrations are equal in each flow component, the amount of material entering the recharge component in a given month is given as

$$Q_r = Q_i \frac{I_r}{I_m} \quad , \quad (B.131)$$

where

- Q_r = radionuclide release entering recharge flow component (g/month),
- Q_t = total radionuclide release from disposal unit (g/month),
- I_r = vertical water percolation rate (cm/month), and
- I_m = total water percolation rate (cm/month).

The mass of material entering the lateral flow component is simply the difference of the total release and the mass of material transported to the aquifer. Annual releases for each flow component are calculated by summing the monthly releases.

B.5 REFERENCES

- Atkinson, A. 1985. *The Time Dependence of pH Within a Repository for Radioactive Waste Disposal*, AERE-R11777, Harwell Laboratory, Oxfordshire, England.
- Atkinson, A., and J. A. Hearne 1984. *An Assessment of the Long-Term Durability of Concrete in Radioactive Waste Repositories*, AERE-R-11465, Harwell Laboratory, Oxfordshire, England.
- Atkinson, A., and J. A. Hearne 1990. "Mechanistic Model for the Durability of Concrete Barriers Exposed to Sulphate-Bearing Groundwaters," *Mat. Res. Soc. Symp. Proc.* **176**, 149–56.
- Baes, C. F. III, and R. D. Sharp 1983. "A Proposal for Estimation of Soil Leaching and Leaching Constants for Use in Assessment Models," *J. Environ. Qual.* **12**(1), 17–28.
- Cowan, H. J. 1982. *Design of Reinforced Concrete Structures*, Prentice-Hall, Englewood Cliffs, N.J.
- Greenberg, S. A., and T. N. Chang 1965. "Investigation of the Colloidal Hydrated Calcium Silicates. II. Solubility Relationships in the Calcium Oxide-Silica-Water System at 25°," *J. Phys. Chem.* **69**, 182–86.
- Hausmann, D. A. 1967. "Steel Corrosion in Concrete," *Materials Protection*, November 19–23.
- Icenhour, A. S. 1995. *Analysis of Source Term Modeling for Low-Level Radioactive Waste Performance Assessments*, ORNL/TM-12908, Lockheed Martin Energy Systems, Inc., Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Icenhour, A. S., and M. L. Tharp 1996. *User's Manual for the SOURCE1 and SOURCE2 Computer Codes: Models for Evaluating Low-Level Radioactive Waste Disposal Facility Source Terms (Version 2.0)*, ORNL/TM-13035, Lockheed Martin Energy Systems, Inc., Oak Ridge National Laboratory, Oak Ridge, Tennessee.
- Jeffrey, G. B. 1920. *Plane Stress and Plane Strain in Bipolar Coordinates*.

- Langelier, W. F. 1936. "The Analytical Control of Anti-Corrosion Water Treatment," *J. Amer. Water Works Assoc.*, 28(10), 1500-21.
- Lea, F. M. 1970. *The Chemistry of Cement and Concrete*, 3rd ed., Edward Arnold, Ltd., London.
- Nawy, E. D. 1966. "Crack Width Control in Welded Fabric Reinforced Centrally Loaded Two-Way Concrete Slabs," presentation at 62nd American Concrete Institute Annual Convention in Philadelphia, March.
- ORNL (Oak Ridge National Laboratory) 1994. *Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6*, ORNL-6783, Martin Marietta Energy Systems, Inc., Oak Ridge, Tennessee.
- Saada, A. S. 1974. *Elasticity Theory and Application*, Pergamon, New York.
- Shuman, R., N. Chau, and E. A. Jennrich 1992. *The SOURCE Computer Codes: Models for Evaluating the Long-Term Performance of SWSA 6 Disposal Units, Version 1.0: User's Manual*, RAE-9005/8-1, Rogers & Associates Engineering Corporation, Salt Lake City, Utah.
- Tuutti, K. 1982. *Corrosion of Steel in Concrete*, Swedish Cement and Concrete Research Institute, Stockholm, Sweden.

Appendix C

TRANSPORT-RELATED PROPERTIES OF RADIONUCLIDES IN WASTE AND CONCRETE USED IN THE PERFORMANCE ASSESSMENT OF SWSA 6 DISPOSAL FACILITIES

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Appendix C

TRANSPORT-RELATED PROPERTIES OF RADIONUCLIDES IN WASTE AND CONCRETE USED IN THE PERFORMANCE ASSESSMENT OF SWSA 6 DISPOSAL FACILITIES

C.1 INTRODUCTION

Modeling radionuclide release rates from a low-level radioactive waste facility involves the use of many parameters, including properties of waste, concrete (the engineered barrier), and pertinent radionuclides. Table C.1 shows many of the necessary transport-related properties used in this performance assessment (PA). Even though there is a large amount of experimental data in the literature, it is rare to find values for all the properties required in a detailed PA. Further, those values found are frequently not at the conditions [e.g., acidity or alkalinity (pH), redox potential (Eh), and concentrations] of interest. If available, data based on accurate inventories, reliable field tests, and carefully controlled laboratory experiments are used in PAs. In the absence of such data, interpolations, extrapolations, and relationships based on experimental results are used. If none of these techniques are available, the procedure used is a correlation based on established theories. A last recourse is empiricism based on established engineering practice. The correlations (based on theories) used in this campaign were selected on the basis of generality and simplicity. The empiricisms used are minor modifications of reliable relationships selected on the basis of overall comprehensiveness.

C.2 BACKGROUND

An understanding of the flux (amount per unit surface area per unit time) from a porous medium requires a knowledge of the properties of the medium that affect transport. The flux of a contaminant is a function of both physical and chemical properties. Physical properties include permeability, porosity, pore distribution, pore geometry, and water content. Chemical properties include those that describe mechanisms for taking contaminants out of or pulling them into the pore solution. These properties range from weak sorption-desorption (e.g., distribution coefficient) to strong chemical bonds (e.g., solubility product constant). With these properties and computer codes (based on appropriate models), predictions can be made of the flux for a contaminant from various

Table C.1. Transport-related properties of radionuclides in waste and concrete used in the performance assessment of SWSA 6 disposal facilities

Constituent	Property	Definition
Waste	Porosity	Volume of voids (air, pore water, etc.) per volume of waste
	Density	Mass of waste per volume of waste
	Chemical type	Inorganic (metal, ceramic, glass, salt, etc.) or organic (cloth, paper, plastic, wood, etc.)
	Nuclide concentration	Amount of nuclide per unit amount of waste
	Ion exchange capacity	Total number of sites available for exchange in the ion exchanger [largely a function of the exchanger (acidic or basic) and the solution pH]
Concrete	Porosity	Volume of voids (air, pore water, etc.) per volume of concrete
	Density	Mass of concrete per volume of concrete
	Composition	Portland cement plus additives (e.g., sand, crushed rock, fly ash, clay, etc.)
	Permeability	Amount of a substance which passes through a material under a given hydraulic pressure (head)
Pore water	pH	Negative log of the hydrogen ion concentration
	Eh	Measured potential difference between an inert electrode and a reference electrode
	Buffer capacity	Response of a system, in terms of pH change, to the addition of hydrogen (H^+) or hydroxide (OH^-) ions
	Complexation capacity	Formation of a complex from a metal ion with another ion by means of one or more chemical bonds
	Ionic strength	A measure of the concentration of an ion and its charge summed over all ions in the solution
	Dielectric constant	The force of attraction between two charges separated by a fixed distance in a uniform medium
Waste/pore water/concrete	Constrictivity	Choking effect that depends on the shape, size, and size distribution of pore space
	Tortuosity	The ratio of the actual tortuous path length for diffusion to the mathematical (i.e., shortest) diffusion path length
	Geometry factor	Tortuosity squared divided by constrictivity
	Distribution coefficient	The distribution or division of a substance between two layers or materials in a definite constant manner (under given conditions)
	Effective diffusion coefficient	A free diffusion coefficient that may be slowed down or speeded up by physical and chemical effects in a heterogeneous system
Radionuclide	Atomic number	Number of protons in the nucleus of an atom

Table C.1. (continued)

Constituent	Property	Definition
	Atomic weight	Total number of neutrons and protons making up the nucleus of an atom
	Atomic (or ionic) radius	Closest distance to which an atom or ion will approach another atom or ion of any size under the conditions specified
	Half-life	Time required for half the atoms of identical radioisotopes to undergo radioactive decay
	Solubility	Maximum amount of a substance that can dissolve in a solvent under the given conditions
	Limiting equivalent ionic conductivity	Contribution made by each ion species of a salt toward an electrolyte's equivalent conductance in the limit of infinite dilution
	Electronegativity	Relative attraction of an atom for the valence electrons in a covalent bond
	Free diffusion coefficient	Diffusion coefficient of an ion in water, or pore liquid, in the limit of infinite dilution

matrices as a function of shape, size, and time. The SOURCE computer codes (SOURCE1 and SOURCE2) used at Oak Ridge National Laboratory (ORNL) (ORNL 1994) for this purpose are examples.

In most cases, values determined by experimentation are not available for the majority of the transport parameters needed to evaluate source terms in the complex systems that most waste disposal facilities represent. In a few cases, values obtained through small-scale experiments and large-scale field tests are available for comparable systems and can be applied with judicious modification. However, in most cases, the needed transport parameters are estimated by the application of equilibrium (e.g., thermodynamic) and kinetic (e.g., diffusion) models to simple heterogeneous systems taken to represent or model the components (e.g., waste and engineered barriers) making up a waste disposal facility. In other words, a transport parameter of a multiphase system is determined from a "mixture law" (relationship or model) that describes its variation with other known or more readily determined properties (measured or estimated from first principles) of the constituents or phases making up the system. This approach to parameter estimation is widely applied to engineering materials that may be considered homogeneously heterogeneous. Examples of such systems include the magnetic properties of composites, the molar heat capacity of solids, and the thermal conductivity of powders.

C.3 TRANSPORT MECHANISMS

In this analysis, the waste and concrete are considered to be unconsolidated and consolidated porous media, respectively. The transport of various species through such media can usually be analyzed and explained in terms of advection and diffusion. In this study, the dominant transport mechanism is assumed to be either advection or diffusion, depending on the leaching time. For example, if there is little to no water movement through the waste (i.e., before failure of engineered barriers), diffusion dominates the radionuclide transport. However, after the concrete cracks and water can contact the porous, grouted waste, advection dominates the transport process. (Note that the SOURCE computer codes used in this PA calculate the release rate by advection and diffusion and compares that value with the solubility limit of the particular species. If the calculated release rate exceeds the solubility limit, then the solubility limit is used to determine the release rate.)

C.3.1 Advection

Release of radionuclides resulting from advection can be modeled using a first-order leaching process accounting for both sorption and decay (Icenhour 1995). Advective leaching is proportional to the amount of water contacting the waste and concrete, and it is inversely proportional to the degree to which the radionuclides are retained by the waste and concrete matrices. For this model, the radionuclide inventory is assumed to be homogeneously mixed within a finite-waste volume that is contacted by a constant infiltration rate of water. The first-order advection equation is given by (Icenhour 1995).

$$L = \frac{\lambda_L}{\lambda_L + \lambda_d} Q_0 [e^{-(\lambda_L + \lambda_d)t_1} - e^{-(\lambda_L + \lambda_d)t_2}] \quad , \quad (C.1)$$

where

- L = mass of radionuclide leached because of advection (g),
- λ_L = leach rate constant (s^{-1}),
- λ_d = radioactive decay constant (s^{-1}),
- Q_0 = initial mass of radionuclide in the waste (g),
- t_1, t_2 = the bounds of the time period of interest (s).

The leach rate constant, λ_L , is given by (Icenhour 1995)

$$\lambda_L = \frac{q}{W\theta R_d} \quad , \quad (C.2)$$

where

- q = water infiltration rate (cm/s),
- W = waste thickness (cm),
- θ = relative saturation (i.e., volume of water in waste/volume of waste) (dimensionless),
- R_d = retardation factor (dimensionless).

Finally, the retardation factor, R_d , can be calculated by the following equation (Icenhour 1995):

$$R_d = 1 + \frac{\rho_b}{\theta} K_d, \quad (C.3)$$

where

- ρ_b = bulk density of waste (g/cm³),
- K_d = distribution coefficient (mL/g).

C.3.2 Diffusion

In addition to advection, the transport of various species through cement-based waste forms can also be analyzed and explained in terms of effective diffusion coefficients (Atkinson, Nelson, and Valentine 1986; Johnston and Wilmot 1992; Godbee et al. 1993). This coefficient for a given species with no solubility constraint can be expressed as (Godbee et al. 1993)

$$D = \frac{D_f}{G(1 + K)H^{-1}}, \quad (C.4)$$

where

- D = effective diffusion coefficient for the species in the porous body (cm²/s),
- D_f = diffusion coefficient of the species in an infinite or free volume of pore liquid (cm²/s),
- G = geometry (or matrix) factor (dimensionless),
- K = distribution coefficient (dimensionless),
- H = relative saturation of pores with liquid (dimensionless).

The geometry factor, G , is defined as τ^2/γ , where τ is tortuosity and γ is constrictivity. Tortuosity is a measure of the diffusion path length in a porous body, and constrictivity is a measure of the choking effect of the pores. In other words, tortuosity is the ratio of the actual tortuous diffusion path length to the mathematical (i.e., shortest) diffusion path length. Similarly, constrictivity is a normalized measure of the filtering and trapping effects of pores. Both factors have sound physical meaning, and independent methods to determine them are available. Tortuosity is a

parameter characterizing the asymmetry of the pore space, whereas constrictivity is a factor that depends on the shape, size, and size distribution of the pore space. Thus, G generally represents a physical containment (retardation) factor of the matrix. The expression $(1 + K)$ can account for phenomena such as sorption/desorption and ion exchange. Thus, $(1 + K)$ represents a chemical containment (retardation) factor. Table C.2 shows the relationship of K to other commonly measured distribution coefficients that have mixed units. Relative pore saturation is the ratio of the water (liquid) volume actually in the pores to the water (liquid) volume in the pores when full. Thus, H represents a retardation factor reflecting the inventory of the transport medium (liquid).

Table C.2. Relationship of the dimensionless distribution coefficient (K) to some other commonly measured distribution coefficients^a

Types of distribution coefficient	Units of distribution coefficient	Relationship of K
Mixed, ^b K_{MP}	$\frac{\text{amount of species/mass of pore-free solid}}{\text{amount of species/volume of liquid}}$	$K = \rho_p \left(\frac{1 - \epsilon}{\epsilon} \right) K_{MP}$
Mixed, ^c K_{MB}	$\frac{\text{amount of species/mass of porous body}}{\text{amount of species/volume of liquid}}$	$K = \rho_b \left(\frac{1}{\epsilon} \right) K_{MB}$
Volume, K_{VV}	$\frac{\text{amount of species/volume of pore-free solid}}{\text{amount of species/volume of liquid}}$	$K = \left(\frac{1 - \epsilon}{\epsilon} \right) K_{VV}$
Geometry, K_{GV}	$\frac{\text{amount of species/volume of porous body}}{\text{amount of species/volume of liquid}}$	$K = \left(\frac{1}{\epsilon} \right) K_{GV}$

^aSource: Godbee et al. 1993.

^b ρ_p is the density of the pore-free solid (i.e., mass of pore-free solid/volume of pore-free solid). ϵ is the void fraction (i.e., volume of pores/volume of porous body).

^c ρ_b is the density of the porous body (mass of porous body/volume of porous body).

If there is a solubility constraint for a species (i.e., the pore liquid can be saturated) the expression $(1 + K)$ is replaced by β^{-2} , where β is defined as the relative saturation concentration of that species in the porous body. The diffusion coefficient for a given species with a solubility constraint can be expressed as (Godbee et al. 1993)

$$D = \frac{D_f}{G(\beta^{-2})H^{-1}} \quad (C.5)$$

The relative solubility constraint is calculated using the relationship (Godbee et al. 1993)

$$\beta = (S_L)(V/A_o)(\epsilon)(\rho)(H) \quad (C.6)$$

where

- S_L = solubility limit of species in pore liquid (g of species/g of pore liquid),
- V = volume of waste form (porous body) (cm^3),
- A_o = initial amount of radionuclide (g),
- ϵ = average open-pore void fraction (dimensionless),
- ρ = density of liquid (g/cm^3).

Species treated as solubility constrained were Be (as BeO), Pd (as PdO), and Pb [as $\text{PbCl}_2 \cdot \text{Pb}(\text{OH})_2$].

When necessary (i.e., data needed for the previous methods are unavailable), corrosion rate data can be used to estimate an effective diffusion coefficient (Godbee 1974) by

$$D = \pi \left[\left(\frac{a_n}{A_o} \right) \left(\frac{V}{S} \right) \left(\frac{1}{t_n} \right) \right]^2 \frac{t_n}{2} \quad (C.7)$$

where

- a_n = amount leached during time interval t_n (g),
- t_n = duration of leaching interval (s),
- S = surface area of sample in contact with leachant (cm^2).

The term in brackets represents a penetration or corrosion rate (length per time). This corrosion-rate approach for estimating a D was used with the metals Al (2 mil per year), Co (1 mil per year), and Cd (3 mil per year).

Speaking broadly, if possible, the distribution coefficients used in Eqs. (C.3) and (C.4) should be determined from leaching tests that determine both a diffusion coefficient and a distribution coefficient, namely, those that approach equilibrium or steady state. Frequently, tests for distribution coefficients that involve grinding the sample and shaking it in a fixed liquid volume (e.g., conventional batch method) or crushing the sample and flowing liquid over it (e.g., conventional column method) often give results that are different from those obtained in diffusion experiments. Among the reasons for this difference is that batch and column tests (because of

crushing, grinding, flow, etc.) may not adequately mimic the surface and pore solution conditions of the porous medium in bulk form.

In this campaign, the distribution coefficients to be used in correlative expressions were obtained from leaching tests carried out at ORNL (Mrochek, Gilliam, and McDaniel 1986) and Pacific Northwest Laboratory (PNL) (Serne et al. 1987). Diffusion (effective) and distribution (dimensionless) coefficients were evaluated from the data using an equation that describes the test configuration (Crank 1993) mathematically (i.e., diffusion into a limited volume of leachant). The expression is of the form (Godbee et al. 1989)

$$\frac{A_t}{A_o - A_f} = 1 - \sum_{n=1}^{\infty} \frac{6\alpha(\alpha + 1) \exp[-Dq_n^2 t / 9(V/S)^2]}{9 + 9\alpha + q_n^2 \alpha^2}, \quad (C.8)$$

where the q_n s are the nonzero roots of

$$\tan q_n = \frac{3q_n}{3 + \alpha q_n^2} \quad (C.9)$$

and

- A_t = total amount leached in time t , mg (μ Ci),
- A_o = initial amount in specimen, mg (μ Ci),
- A_f = amount in specimen at equilibrium, mg (μ Ci),
- D = effective diffusion coefficient (cm^2/s),
- t = time (s),
- V = volume of specimen (cm^3),
- S = surface area (geometric) of specimen (cm^2),
- $\alpha = \frac{V_L}{\epsilon KV} = \frac{A_o - A_f}{A_f}$ (dimensionless),
- V_L = volume of leachant (cm^3),
- ϵ = void fraction (porosity) (dimensionless),
- K = distribution coefficient (dimensionless).

Rearrangement of the expressions given for α above shows that

$$K = \frac{\left(\frac{A_f}{V}\right)}{\epsilon \left(\frac{A_o - A_f}{V_L}\right)} \quad (C.10)$$

The parameters A_0 , V , S , and ϵ are properties of the specimen, whereas V_L is a property of the leachant. All are determined at the beginning of the experiment. The parameter A_t is a property of the system and is generated during the experiment as a function of time t . The specimen properties D and A_f are evaluated by fitting the data $[A_t(t)]$ to Eq. (C.8). A value for K is then obtained using Eq. (C.10).

The ORNL and PNL experimental data cited above were fitted to Eq. (C.8) using the computer program NEWBOX (Nestor, Godbee, and Joy, in press) developed at ORNL. NEWBOX, which is executable on a personal computer, calculates the fractional release of material from four different geometrical shapes (semi-infinite medium; finite slab; finite, circular cylinder; and sphere) and accounts for several different boundary conditions.

The solution of mass transport equations (e.g., Fick's first and second laws), subject to appropriate initial and boundary conditions, are usually nonlinear in some parameters (diffusion coefficient, reaction rate constants, etc.). Thus, use of a method of parameter adjustment involving first partial derivatives can be complicated and prone to errors in the computation of the derivatives. In addition, the parameters must satisfy certain constraints; for example, the diffusion coefficient must remain positive. For these reasons, a variant of the constrained simplex method (Box, Davies, and Swann 1969) is used in NEWBOX to estimate parameters. It is similar, but not identical, to the downhill simplex method (Nelder and Mead 1965). In general, NEWBOX calculates the fraction of material transferred as a function of time from an expression obtained by the inversion (Abramowitz and Stegun 1964) of the Laplace transform of the fraction transferred, rather than by taking derivatives of a calculated concentration profile.

Under certain conditions, some nuclides (e.g., Pd, Th, and Zr) may have low solubilities such that they are transported (by both advection and diffusion) at a rate which is limited by their solubilities. If the pore liquid concentration of a species is at its solubility limit concentration, the effective diffusion coefficient D is calculated using Eq. (C.5). In this case, the relative solubility constraint β is evaluated with Eq. (C.6) using solubility limit data from the literature.

In many cases, a geometry factor G [e.g., in Eqs. (C.4) and (C.5)] is not available for a nuclide of concern in the environment of interest. In this campaign, a semiempirical expression is used for estimating geometry factors of related (i.e., similar chemical and physical characteristics) nuclides in the same porous medium, namely (Godbee 1996),

$$G_x = G_a \left(\frac{r_x}{r_a} \right)^2, \quad (C.11)$$

where

- G_x = geometry (matrix) factor of nuclide to be estimated (dimensionless),
- G_a = geometry factor of known similar (e.g., chemical group, oxidation state, etc.) nuclide (dimensionless),
- r_x = ionic radius of x (Å),

r_a = ionic radius of a (Å).

The relationship is derived largely from an analysis of the data in Mrochek, Gilliam, and McDaniel (1986) and Serne et al. (1987) when fitted to Eq. (C.8).

An empirical correlation is used for estimating geometry factors of the same nuclide in related (i.e., comparable physical retardation factors) porous media. The expression is based on one of the earliest and simplest concepts to relate geometry factor and porosity (Brown et al. 1950), namely,

$$G_x = G_a \left(\frac{\epsilon_a}{\epsilon_x} \right)^2 \left(\frac{1 - \epsilon_x}{1 - \epsilon_a} \right), \quad (C.12)$$

where

ϵ_a = porosity of medium a (dimensionless),

ϵ_x = porosity of medium x (dimensionless).

Often a distribution coefficient K is not available for a nuclide of concern in the environment of interest. In a manner similar to that used with G, an unsophisticated semiempirical expression [based on an analysis of the data in Mrochek, Gilliam, and McDaniel (1986) and Serne et al. (1987), coupled with the rules for ionic selectivity summarized in Lin (1973)] is used to estimate the needed distribution coefficient from that of a similar nuclide (i.e., same chemical family or group, oxidation state, etc.). The form of the relationship is

$$K_x = K_a \left(\frac{Z_x}{Z_a} \right), \quad (C.13)$$

where

K_x = distribution coefficient of nuclide to be estimated (dimensionless),

K_a = distribution coefficient of known similar nuclide (dimensionless),

Z_x = atomic number of nuclide x (dimensionless),

Z_a = atomic number of nuclide a (dimensionless).

A correlation that allows the distribution coefficient of a given nuclide in a porous medium to be transposed to a different (but comparable) porous medium is required. The elementary relationship used in this campaign is

$$K_x = K_a \left(\frac{1 - \epsilon_x}{1 - \epsilon_a} \right) \left(\frac{I_x}{I_a} \right), \quad (\text{C.14})$$

where

- K_x = distribution coefficient of nuclide to be estimated (dimensionless),
- K_a = distribution coefficient of nuclide in medium a (dimensionless),
- ϵ_x = porosity of medium x (dimensionless),
- ϵ_a = porosity of medium a (dimensionless),
- I_x = number (n) of ion exchange sites per volume of medium x (n/cm^3),
- I_a = number (n) of ion exchange sites per volume of medium a, (n/cm^3).

In this campaign, largely because of the dearth of information, a value of 1.0 for I_x / I_a is considered to be reasonable, except for the alkali metals, in which case it is assumed to be proportional to the average pH of the two media.

The free-diffusion coefficient D_f , sometimes referred to as the self-diffusion coefficient (Godbee 1996), is a property of the nuclide in the pore liquid. In other words, it is the diffusion coefficient of the nuclide in the pore liquid if the nuclide is free of the constraints imposed by surfaces, ionic (molecular) filtration, dead-end pores, etc. The free-diffusion coefficients in this campaign were taken from an embryonic database on mass transport parameters compiled at ORNL.* The majority of the D_f values in this database were obtained from the literature. However, when not readily available, they are calculated* using the Nernst-Einstein equation (Atkins 1990), namely,

$$D_f = \left(\frac{RT}{F^2} \right) \left(\frac{\lambda}{z} \right), \quad (\text{C.15})$$

where

- R = gas constant ($8.3144 \text{ J K}^{-1} \text{ mol}^{-1}$),
- T = absolute temperature (K),
- F = Faraday constant ($9.6493 \times 10^4 \text{ C/mol}$),
- λ = equivalent ionic conductance ($\text{cm}^2 \text{ S mol}^{-1}$),
- z = charge number of the ion.

*S. L. Loghry, Oak Ridge Natl. Lab., "Property Summary," correspondence to H. W. Godbee, Delta-21 Resources, Inc., Oak Ridge, Tenn., July 7, 1995.

In Eq. (C.15), note that the elementary entity used in a mole (mol) is an electron. Values for λ are available in handbooks on physical chemistry, the literature on electrochemistry, and the ORNL database previously mentioned.

In this campaign, the value of H (relative saturation of the pores) is set equal to 1.0. In other words, the pores are full of liquid. Saturation is usually assumed in PAs so that a predicted release is conservative (i.e., higher than if $H < 1.0$).

C.4 SUMMARY AND CONCLUSIONS

The direct results of the methodology described in this appendix are estimates of the needed diffusion and distribution coefficients for the radionuclides considered in this PA. Further detail on this step-by-step procedure to estimate mass transport parameters when experimental data are not available can be found in Godbee (1996). In summary, this methodology basically involves the use of three kinds of relationships (arranged in a hierarchy of preference):

1. expressions based on established theories that relate a parameter to more fundamental physicochemical properties of the materials making up the system [e.g., Eq. (C.15) for D_f];
2. correlations based on the statistically meaningful analyses of reliable experimental data on similar systems [e.g., Eq. (C.13) for K]; and
3. empiricisms based on engineering judgment derived from experience with related applications [e.g., Eq. (C.11) for G].

The estimated transport parameters used in the SWSA 6 PA are presented in Table C.3, Table C.4, and Table C.5. Calculations for some elements are detailed in Godbee (1996). The methodology outlined in this appendix was used in developing all of the parameters presented in Table C.3, Table C.4, and Table C.5. The estimated values in these tables appear to be reasonable and should be applicable to assessment studies in the absence of system-specific experimental results (compare Serne et al. 1993). These conclusions are supported largely by the following:

- The values are within the range of experimental results available in the literature for such (or comparable) species in porous media.
- The values are based on representative and internally consistent ancillary parameters that are combined using relationships that range from those grounded in established theories to those rooted in defensible empiricism.
- The values are recognized as estimates of parameters that have unequivocal meaning only in the system in which they are intended to be surrogates for experimental results.

Table C.3. Distribution coefficients and solubility limits of elements evaluated in the SWSA 6 performance assessment

Element	Formula	Solubility ^a (mol/L)	Waste distribution coefficient ^b		Concrete distribution coefficient ^b	
			K	K _d	K	K _d
			(dimensionless)	(mL/g)	(dimensionless)	(mL/g)
H	T ₂ O	1.11×10^2	1.00×10^0	1.99×10^{-1}	1.31×10^0	8.53×10^{-2}
Be	BeO	8.00×10^{-6}	1.57×10^7	3.12×10^6	6.05×10^0	3.93×10^{-1}
C	BaCO ₃	1.11×10^{-4}	5.50×10^0	1.09×10^0	7.19×10^0	4.68×10^{-1}
Al	Al	5.24×10^0	8.45×10^5	1.68×10^5	2.50×10^1	1.63×10^0
Cl	KCl	4.61×10^0	8.88×10^{-1}	1.77×10^{-1}	1.16×10^0	7.55×10^{-2}
K	KCl	4.61×10^0	3.59×10^1	7.14×10^0	6.91×10^1	4.50×10^0
Ca	CaCO ₃	1.53×10^{-4}	2.32×10^1	4.61×10^0	3.02×10^1	1.97×10^0
Co	Co	3.46×10^0	1.04×10^7	2.07×10^6	4.05×10^1	2.64×10^0
Ni	NiCO ₃	7.83×10^{-4}	3.21×10^1	6.38×10^0	4.20×10^1	2.73×10^0
Se	NaSeO ₄	4.45×10^0	1.11×10^1	2.21×10^1	1.45×10^1	9.44×10^{-1}
Rb	Rb ₂ CO ₃	1.95×10^1	7.02×10^1	1.40×10^1	1.35×10^2	8.79×10^0
Sr	SrCO ₃	7.45×10^{-5}	4.40×10^1	8.75×10^0	5.75×10^1	3.74×10^0
Zr	Zr(OH) ₄	2.19×10^{-3}	2.50×10^2	4.97×10^1	3.27×10^2	2.13×10^1
Nb	NbOF ₃ ·2KF	2.60×10^{-1}	2.56×10^2	5.90×10^1	3.35×10^2	2.18×10^1
Tc	NH ₄ TcO ₄	9.12×10^{-2}	1.60×10^1	3.18×10^0	2.09×10^1	1.36×10^0
Pd	PdO	3.76×10^{-9}	7.00×10^6	1.39×10^6	6.90×10^1	4.49×10^0
Cd	Cd	7.64×10^0	7.17×10^5	1.42×10^5	7.20×10^1	4.69×10^0
Sn	SnCl ₂	1.37×10^1	1.52×10^2	3.02×10^1	1.99×10^2	1.30×10^1
I	KI	7.68×10^0	2.77×10^0	5.51×10^{-1}	3.62×10^0	2.36×10^{-1}
Cs	Cs ₂ CO ₃	1.60×10^1	1.00×10^2	1.99×10^1	2.00×10^2	1.30×10^1
Sm	Sm(Cl ₃)	3.60×10^0	3.12×10^2	6.20×10^1	4.08×10^2	2.66×10^1
Eu	Eu ₂ O ₃	2.84×10^{-6}	3.17×10^2	6.30×10^1	4.15×10^2	2.72×10^1
Os	OsO ₄	2.24×10^{-1}	1.73×10^2	3.44×10^1	1.23×10^3	8.01×10^1
Pb	PbCl ₂ ·Pb(OH) ₂	5.16×10^{-5}	2.49×10^2	4.95×10^1	3.26×10^2	2.12×10^1
Bi	Bi(OH) ₃	5.40×10^{-6}	2.41×10^1	4.79×10^0	3.15×10^1	2.05×10^0
Ra	RaSO ₄	6.21×10^{-8}	1.02×10^2	2.03×10^1	1.33×10^2	8.66×10^0
Ac	Ac ₂ O ₃	1.55×10^{-5}	1.85×10^4	3.69×10^3	2.42×10^4	1.55×10^3
Ac ^c	Ac ₂ O ₃	1.55×10^{-5}	2.67×10^2	5.31×10^1	3.49×10^2	2.27×10^1
Th	ThO ₂	3.00×10^{-9}	1.87×10^4	3.73×10^3	2.45×10^4	1.57×10^3
Th ^c	ThO ₂	7.57×10^{-8}	2.70×10^2	5.37×10^1	3.53×10^2	2.30×10^1
Pa	PaO ₂	5.71×10^{-4}	1.89×10^4	3.77×10^3	2.48×10^4	1.59×10^3
Pa ^c	PaO ₂	5.71×10^{-4}	2.73×10^2	5.43×10^1	3.57×10^2	2.32×10^1
U	UO ₂	3.00×10^{-10}	1.92×10^4	3.82×10^3	2.51×10^4	1.61×10^3
U ^c	UO ₂	1.46×10^{-6}	2.76×10^2	5.49×10^1	3.61×10^2	2.35×10^1
Np	Np ₂ O ₃	3.00×10^{-9}	1.94×10^4	3.85×10^3	2.53×10^4	1.62×10^3

Table C.3. (continued)

Element	Formula	Solubility ^a (mol/L)	Waste distribution coefficient ^b		Concrete distribution coefficient ^b	
			K (dimensionless)	K _d (mL/g)	K (dimensionless)	K _d (mL/g)
Np ^c	Np ₂ O ₃	4.05 × 10 ⁻⁵	2.79 × 10 ²	5.55 × 10 ¹	3.65 × 10 ²	2.38 × 10 ¹
Pu	PuO ₂	1.00 × 10 ⁻¹⁰	1.96 × 10 ⁴	3.90 × 10 ³	2.56 × 10 ⁴	1.64 × 10 ³
Pu ^c	PuO ₂	1.71 × 10 ⁻⁸	2.82 × 10 ²	5.61 × 10 ¹	3.69 × 10 ²	2.40 × 10 ¹
Am	Am(OH) ₃	4.00 × 10 ⁻¹⁰	1.98 × 10 ⁴	3.93 × 10 ³	2.59 × 10 ⁴	1.66 × 10 ³
Am ^c	Am ₂ O ₃	3.27 × 10 ⁻⁶	2.85 × 10 ²	5.67 × 10 ¹	3.73 × 10 ²	2.43 × 10 ¹
Cm	Cm(OH) ₃	1.00 × 10 ⁻¹⁰	2.00 × 10 ⁴	3.98 × 10 ³	2.61 × 10 ⁴	1.67 × 10 ³
Cm ^c	Cm ₂ O ₃	2.84 × 10 ⁻⁶	2.88 × 10 ²	5.73 × 10 ¹	3.77 × 10 ²	2.45 × 10 ¹
Cf	Cf(NO ₃) ₃	3.49 × 10 ⁰	2.04 × 10 ⁴	4.06 × 10 ³	2.64 × 10 ⁴	1.69 × 10 ³
Cf ^c	Cf(NO ₃) ₃	3.49 × 10 ⁰	2.91 × 10 ²	5.79 × 10 ¹	3.81 × 10 ²	2.48 × 10 ¹

^aSources: D. R. Lide, ed., *CRC Handbook of Chemistry and Physics*, 73rd ed., CRC Press, Boca Raton, Fla., 1992; Freie Universität Berlin and Institut für Anorganische und Analytische Chemie, *Solubility and Speciation of Actinides in Salt Solutions and Migration Experiments of Intermediate Level Waste in Salt Formations*, FUB/FI 53132-415/85, 1986; M. Schweingruber, *Actinide Solubility in Deep Groundwaters—Estimates for Upper Limits Based on Chemical Equilibrium Calculations*, EIR-Bericht Nr. 507, Eidgenössisches Institut für Reaktorforschung, Würenlingen, Switzerland, 1983; Agence Internationale de l'Energie Atomique, *Seminaire sur les Techniques d'Etude et les Methodes d'Evaluation des Sites en vue du Stockage Definitif Souterrain des Dechets Radioactifs*, IAEA-SR-104/25, Vienna, 1984; I. G. McKinley and W. R. Alexander, "A Review of the Use of Natural Analogues to Test Performance Assessment Models of a Cementitious Near Field," *Waste Management* 12, 253-59 (1992); N. A. Chapman and R. H. Flowers, "Near Field Solubility Constraints on Radionuclide Mobilization and Their Influence on Waste Package Design," *Philosophical Transactions of the Royal Society of London*, 319, 83-95 (1986).

^bSources: H. A. Friedman and A. D. Kelmers, *Laboratory Measurement of Radionuclide Sorption in Solid Waste Storage Area 6 Soil/Groundwater Systems*, ORNL-TM-10561, Martin Marietta Energy Systems, Inc., Oak Ridge National Laboratory, Oak Ridge, Tenn., June 1990; I. Neretnieks, "Diffusivities of Some Constituents in Compacted Wet Bentonite Clay and the Impact on Radionuclide Migration in the Buffer," *Nuclear Technology* 71, 458-70 (1985); J. G. Moore, H. W. Godbee, A. H. Kibbey, and D. S. Joy., *Development of Cementitious Grouts for the Incorporation of Radioactive Wastes. Part I Leach Studies*, ORNL-4962, Oak Ridge National Laboratory, Oak Ridge, Tenn., April 1975.

^cValues used for mixtures of actinide wastes and soils (i.e., trenches). In trenches, the pH is not expected to be basic; therefore, more conservative transport parameters (for the actinides) based on ORNL (1994) are used.

Table C.4. Transport parameters used to calculate diffusion coefficients for waste (D_w)

Element	Free-diffusion coefficient D_f (m^2/s) ^b	Transport parameters (dimensionless) for D_w ^a			Waste diffusion coefficient D_w (m^2/s)
		Geometry factor G	Distribution coefficient K	Retardation factor R	
H	9.31×10^{-9}	1.20×10^0	1.00×10^0	2.40×10^0	3.88×10^{-9}
Be	5.99×10^{-10}	1.05×10^0	1.57×10^7	1.65×10^7	3.63×10^{-17}
C	9.20×10^{-10}	9.12×10^0	5.50×10^0	5.93×10^1	1.55×10^{-11}
Al	5.41×10^{-10}	5.00×10^0	8.45×10^5	4.23×10^6	1.28×10^{-16}
Cl	2.03×10^{-9}	1.48×10^1	8.88×10^{-1}	2.79×10^1	7.28×10^{-11}
K	1.96×10^{-9}	1.83×10^0	3.59×10^1	6.75×10^1	2.90×10^{-11}
Ca	7.92×10^{-10}	1.02×10^1	2.32×10^1	2.47×10^2	3.12×10^{-12}
Co	7.32×10^{-10}	2.20×10^0	1.04×10^7	2.29×10^7	3.21×10^{-17}
Ni	6.60×10^{-10}	1.99×10^0	3.21×10^1	6.59×10^1	1.00×10^{-11}
Se	1.18×10^{-9}	7.30×10^0	1.11×10^1	8.83×10^1	1.34×10^{-11}
Rb	2.07×10^{-9}	2.29×10^0	7.02×10^1	1.63×10^2	1.27×10^{-11}
Sr	7.91×10^{-10}	1.47×10^1	4.40×10^1	6.62×10^2	1.19×10^{-12}
Zr	7.92×10^{-10}	7.90×10^0	2.50×10^2	1.98×10^3	4.00×10^{-13}
Nb	4.84×10^{-10}	4.97×10^0	2.56×10^2	1.28×10^3	3.78×10^{-13}
Tc	1.50×10^{-9}	3.50×10^1	1.60×10^1	5.95×10^2	2.52×10^{-12}
Pd	7.50×10^{-10}	2.42×10^0	7.00×10^6	1.69×10^7	4.44×10^{-17}
Cd	7.19×10^{-10}	3.47×10^0	7.17×10^5	2.49×10^6	2.89×10^{-16}
Sn	7.05×10^{-10}	2.05×10^1	1.52×10^2	3.14×10^3	2.25×10^{-13}
I	2.05×10^{-9}	2.18×10^1	2.77×10^0	8.22×10^1	2.49×10^{-11}
Cs	2.06×10^{-9}	2.81×10^0	1.00×10^2	2.84×10^2	7.26×10^{-12}
Sm	6.08×10^{-10}	3.90×10^1	3.12×10^2	1.22×10^4	4.98×10^{-14}
Eu	6.02×10^{-10}	3.75×10^1	3.17×10^2	1.19×10^4	5.06×10^{-14}
Pb	9.45×10^{-10}	4.12×10^1	2.49×10^2	1.03×10^4	9.17×10^{-14}
Bi	8.10×10^{-10}	1.53×10^1	2.41×10^1	3.84×10^2	2.11×10^{-12}
Ra	8.89×10^{-10}	2.10×10^1	1.02×10^2	2.16×10^3	4.12×10^{-13}
Ac	4.59×10^{-10}	8.24×10^1	1.85×10^4	1.52×10^6	3.01×10^{-16}
Ac ^c	4.59×10^{-10}	8.24×10^1	2.67×10^2	2.21×10^4	2.08×10^{-14}
Th	5.10×10^{-10}	5.80×10^1	1.87×10^4	1.08×10^6	4.70×10^{-16}
Th ^c	5.10×10^{-10}	5.80×10^1	2.70×10^2	1.57×10^4	3.25×10^{-14}
Pa	5.24×10^{-10}	5.68×10^1	1.89×10^4	1.07×10^6	4.88×10^{-16}
Pa ^c	5.24×10^{-10}	5.68×10^1	2.73×10^2	1.56×10^4	3.36×10^{-14}
U	5.61×10^{-10}	5.57×10^1	1.92×10^4	1.07×10^6	5.25×10^{-16}
U ^c	5.61×10^{-10}	5.57×10^1	2.76×10^2	1.54×10^4	3.64×10^{-14}
Np	5.61×10^{-10}	7.16×10^1	1.94×10^4	1.39×10^6	4.04×10^{-16}
Np ^c	5.61×10^{-10}	7.16×10^1	2.79×10^2	2.00×10^4	2.81×10^{-14}
Pu	5.61×10^{-10}	5.12×10^1	1.96×10^4	1.00×10^6	5.59×10^{-16}
Pu ^c	5.61×10^{-10}	5.12×10^1	2.82×10^2	1.45×10^4	3.87×10^{-14}

Table C.4. (continued)

Element	Free-diffusion coefficient D_f (m^2/s) ^b	Transport parameters (dimensionless) for D_w ^a			Waste diffusion coefficient D_w (m^2/s)
		Geometry factor G	Distribution coefficient K	Retardation factor R	
Am	5.10×10^{-10}	6.78×10^1	1.98×10^4	1.34×10^6	3.80×10^{-16}
Am ^c	5.10×10^{-10}	6.78×10^1	2.85×10^2	1.94×10^4	2.63×10^{-14}
Cm	5.10×10^{-10}	5.80×10^1	2.00×10^4	1.16×10^6	4.40×10^{-16}
Cm ^c	5.10×10^{-10}	5.80×10^1	2.88×10^2	1.68×10^4	3.04×10^{-14}
Cf	5.10×10^{-10}	5.68×10^1	2.04×10^4	1.16×10^6	4.40×10^{-16}
Cf ^c	5.10×10^{-10}	5.68×10^1	2.91×10^2	1.66×10^4	3.07×10^{-14}

^a $D_w = D_f/R$ where $R = G(1 + K)H^{-1}$. The term H is defined as the relative saturation of the pores with water. The pores are assumed to be full in the calculations shown in this table; that is, $H = 1.0$.

^bSources: D. G. Miller, *Estimation of Tracer Diffusion Coefficients of Ions in Aqueous Solution*, UCRL-53319, Lawrence Livermore Laboratory, University of California, Livermore, Calif., 1982; C. J. Geankoplis, *Mass Transport Phenomena*, Holt, Rinehart and Winston, New York, 1972; F. Kepak, "Adsorption and Colloidal Properties of Radioactive Elements in Trace Concentrations," *Chemical Review* 71(4), 1972; E. L. Cussler, *Diffusion—Mass Transfer in Fluid Systems*, Cambridge University Press, New York, 1984; *CRC Handbook of Chemistry and Physics*, ed. D. R. Lide, 74th ed., CRC Press, Boca Raton, Fla., 1993; *Lange's Handbook of Chemistry*, ed. J. A. Dean, 14th ed., McGraw-Hill, Inc., New York, 1992.

^cValues used for mixtures of actinide wastes and soils (i.e., trenches). In trenches, the pH is not expected to be basic; therefore, more conservative transport parameters (for the actinides) based on ORNL (1994) are used

Table C.5. Transport parameters used to calculate diffusion coefficients for concrete (D_c)

Element	Free-diffusion coefficient D_f (m^2/s) ^b	Transport parameters (dimensionless) for D_c ^a			Concrete diffusion coefficient D_c (m^2/s)
		Geometry factor G	Distribution coefficient K	Retardation factor R	
H	9.31×10^{-9}	8.54×10^0	1.31×10^0	1.97×10^1	4.73×10^{-10}
Be	5.99×10^{-10}	7.48×10^0	6.05×10^0	5.27×10^1	1.14×10^{-11}
C	9.20×10^{-10}	6.50×10^1	7.19×10^0	5.33×10^2	1.73×10^{-12}
Al	5.41×10^{-10}	3.56×10^1	2.50×10^1	9.26×10^2	5.84×10^{-13}
Cl	2.03×10^{-9}	1.05×10^2	1.16×10^0	2.26×10^2	8.98×10^{-12}
K	1.96×10^{-9}	1.30×10^1	6.91×10^1	9.11×10^2	2.15×10^{-12}
Ca	7.92×10^{-10}	7.26×10^1	3.02×10^1	2.27×10^3	3.49×10^{-13}
Co	7.32×10^{-10}	1.57×10^1	4.05×10^1	6.52×10^2	1.12×10^{-12}
Ni	6.60×10^{-10}	1.42×10^1	4.20×10^1	6.11×10^2	1.08×10^{-12}
Se	1.18×10^{-9}	5.20×10^1	1.45×10^1	8.06×10^2	1.46×10^{-12}
Rb	2.07×10^{-9}	1.63×10^1	1.35×10^2	2.22×10^3	9.32×10^{-13}
Sr	7.91×10^{-10}	1.05×10^2	5.75×10^1	6.14×10^3	1.29×10^{-13}
Zr	7.92×10^{-10}	5.62×10^1	3.27×10^2	1.84×10^4	4.30×10^{-14}
Nb	4.84×10^{-10}	3.54×10^1	3.35×10^2	1.19×10^4	4.07×10^{-14}
Tc	1.50×10^{-9}	2.49×10^2	2.09×10^1	5.45×10^3	2.75×10^{-13}
Pd	7.50×10^{-10}	1.72×10^1	6.90×10^1	1.20×10^3	6.25×10^{-13}
Cd	7.19×10^{-10}	2.47×10^1	7.20×10^1	1.80×10^3	3.99×10^{-13}
Sn	7.05×10^{-10}	1.46×10^2	1.99×10^2	2.92×10^4	2.41×10^{-14}
I	2.05×10^{-9}	1.55×10^2	3.62×10^0	7.16×10^2	2.86×10^{-12}
Cs	2.06×10^{-9}	2.00×10^1	2.00×10^2	4.02×10^3	5.12×10^{-13}
Sm	6.08×10^{-10}	2.78×10^2	4.08×10^2	1.14×10^5	5.33×10^{-15}
Eu	6.02×10^{-10}	2.70×10^2	4.15×10^2	1.12×10^5	5.38×10^{-15}
Pb	9.45×10^{-10}	2.93×10^2	3.26×10^2	9.58×10^4	9.86×10^{-15}
Bi	8.10×10^{-10}	1.09×10^2	3.15×10^1	3.54×10^3	2.29×10^{-13}
Ra	8.89×10^{-10}	1.50×10^2	1.33×10^2	2.01×10^4	4.42×10^{-14}
Ac	4.59×10^{-10}	5.87×10^2	2.42×10^4	1.42×10^7	3.23×10^{-17}
Th	5.10×10^{-10}	4.13×10^2	2.45×10^4	1.01×10^7	5.04×10^{-17}
Pa	5.24×10^{-10}	4.04×10^2	2.48×10^4	1.00×10^7	5.23×10^{-17}
U	5.61×10^{-10}	3.97×10^2	2.51×10^4	9.97×10^6	5.63×10^{-17}
Np	5.61×10^{-10}	5.10×10^2	2.53×10^4	1.29×10^7	4.35×10^{-17}
Pu	5.61×10^{-10}	3.65×10^2	2.56×10^4	9.34×10^6	6.00×10^{-17}
Am	5.10×10^{-10}	4.83×10^2	2.59×10^4	1.25×10^7	4.08×10^{-17}
Cm	5.10×10^{-10}	4.13×10^2	2.61×10^4	1.08×10^7	5.18×10^{-17}
Cf	5.10×10^{-10}	4.04×10^2	2.64×10^4	1.07×10^7	5.01×10^{-17}

^a $D_c = D_f/R$ where $R = G(1 + K)H^{-1}$. The term H is defined as the relative saturation of the pores with water. The pores are assumed to be full in the calculations shown in this table; that is, $H = 1.0$.

^bSources: D. G. Miller, *Estimation of Tracer Diffusion Coefficients of Ions in Aqueous Solution*, UCRL-53319, Lawrence Livermore Laboratory, University of California, Livermore, Calif., 1982; C. J. Geankoplis, *Mass Transport Phenomena*, Holt, Rinehart and Winston, New York, 1972; F. Kepak, "Adsorption and Colloidal Properties of Radioactive Elements in Trace Concentrations," *Chemical Review* 71(4), 1972; E. L. Cussler, *Diffusion—Mass Transfer in Fluid Systems*, Cambridge University Press, New York, 1984; *CRC Handbook of Chemistry and Physics*, ed. D. R. Lide, 74th ed., CRC Press, Boca Raton, Fla., 1993; *Lange's Handbook of Chemistry*, ed. J. A. Dean, 14th ed., McGraw-Hill, Inc., New York, 1992.

C.5 REFERENCES

- Abramowitz, M., and I. E. Stegun 1964. *Handbook of Mathematical Functions*, National Bureau of Standards, Bethesda, Md.
- Atkins, P. W. 1990. *Physical Chemistry*, 4th ed., W. H. Freeman and Co., New York, p. 765.
- Atkinson, A., K. Nelson, and T. M. Valentine 1986. "Leach Test Characterization of Cement-Based Nuclear Waste Forms," *Nuclear Chemical Waste Management* 6, 241-53.
- Box, M. J., D. Davies, and W. H. Swann 1969. *Non-linear Optimization Techniques*, Oliver and Boyd, London.
- Brown, G. G. et al. 1950. *Unit Operations*, John Wiley & Sons, New York, pp. 217-18.
- Crank, J. 1993. *The Mathematics of Diffusion*, 2nd ed., Oxford University Press, Oxford, pp. 93-96.
- Godbee, H. W. 1996. *Estimated Transport Parameters for Selected Radionuclides in Low-Level Waste and Concrete Containment*, D21-A20005-01-95, Delta-21 Resources, Inc., Knoxville, Tenn.
- Godbee, H. W., and D. S. Joy, 1974, *Assessment of the Loss of Radioactive Isotopes from Waste Solids to the Environment. Part I: Background and Theory*, ORNL/TM-4333, Union Carbide Corp., Oak Ridge Natl. Lab.
- Godbee, H. W. et al. 1989. "Application of the NEWBOX Computer Program to an Analysis of the Leaching Data for Several Alkali Metals and Phenol from Cementitious Waste Forms," pp. 184-188 in preprints of papers presented at the 197th ACS National Meeting, Dallas, Tex., April 9-14.
- Godbee, H. W. et al. 1993. "Waste Confinement Systems and Waste-Form Durability," pp. 125-41 in *Effective and Safe Waste Management: Interfacing Sciences and Engineering with Monitoring and Risk Analysis*, ed. R. L. Jolley and R. G. M. Wang, Lewis Publishers, Boca Raton, Fla.
- Icenhour, A. S. 1995. *Analysis of Source Term Modeling for Low-Level Radioactive Waste Performance Assessments*, ORNL/TM-12908, Lockheed Martin Energy Systems, Inc., Oak Ridge Natl. Lab.
- Johnston, H. M., and D. J. Wilmot 1992. "Sorption and Diffusion Studies in Cementitious Grouts," *Waste Management* 12, 289-97.
- Lin, K. H. 1973. *Use of Ion Exchange for the Treatment of Liquids in Nuclear Power Plants*, ORNL-4792, Union Carbide Corp., Oak Ridge Natl. Lab.
- Mrochek, J. E., T. M. Gilliam, and E. W. McDaniel 1986. "Grout Formulation for the Immobilization of Mixed Organic Containing Wastes," pp. 55-76 in *American Nuclear Society International*

Topical Meeting on Waste Management and Decontamination and Decommissioning, proceedings of SPECTRUM '86, Niagara Falls, N.Y., Sept. 14-18.

Nelder, J. A., and R. Mead 1965. "A Simplex Method for Function Minimization," *Computing Journal* 7, 308-13.

Nestor, C. W., Jr., H. W. Godbee, and D. S. Joy. *NEWBOX: A Computer Program for Parameter Estimation in Diffusion Problems*, ORNL/TM-10910, Lockheed Martin Energy Research Corp., Oak Ridge Natl. Lab. (in press).

ORNL (Oak Ridge National Laboratory) 1994. *Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6*, ORNL-6783, Martin Marietta Energy Systems, Inc., Oak Ridge Natl. Lab.

Serne, R. J. et al. 1987. *Laboratory Leach Tests of Phosphate/Sulfate Waste Grout and Leachate Adsorption Tests Using Hanford Sediment*, PNL-6019, Pacific Northwest Laboratory Richland, Wash.

Serne, R. J. et al. 1993. *Solid-Waste Leach Characteristics and Contaminant-Sediment Interactions Volume 1: Batch Leach and Adsorption Tests and Sediment Characterization*, PNL-8889 Vol. 1, Pacific Northwest Laboratory, Richland, Wash.

Appendix D

HYDROLOGY AND SUBSURFACE TRANSPORT MODELING

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Appendix D

HYDROLOGY AND SUBSURFACE TRANSPORT MODELING

This appendix contains subsections describing technical aspects of terrain analysis, the Unified Transport Model (UTM), and the PADSIM and HOLSIM codes.

D.1 TERRAIN ANALYSIS

D.1.1 Introduction

Land surface topography has a large influence on water flow within and through a watershed. Current land surfaces reflect water flow and erosion processes determined by interactions between climate, vegetation, soil profiles, and bedrock. Methods for incorporating the topographic features of watersheds into hydrologic modeling were initiated by Beven and Kirkby (1979) and have been applied extensively in hydrologic research (e.g., Quinn et al. 1991). Application of these methods for hydrologic characterization of SWSA 6 was undertaken.

The movement of water into and over a watershed may lead to stream flow caused by hydrologic processes that are significantly affected by

- the intensity and duration of precipitation,
- the porosity and permeability of soil and bedrock in the watershed,
- the antecedent water storage in the watershed, and
- the shape of the landform that gathers water flow to the stream bed.

Change in soil water storage (wetness) at a particular place is a function of the rates of water arrival and departure. As soil water storage increases toward saturation, overland flow may develop. Surface water will collect or disperse depending upon topography. For example, where contours converge, as in a stream valley, water will collect favoring the development of saturated areas; where contours diverge, as on ridges, water will disperse (toward convergent flow zones), reducing the tendency for soil saturation. The combination of basin convergence or divergence and land slope can be used to derive a spatial distribution of topographic index. This index describes the propensity for each place in a watershed to become saturated. Topographic index is of significant value for its use in modeling the hydrologic behavior of a watershed in response to precipitation. Simulated flow may be

calibrated with stream flow measurements, and additional flow measurements may be used for model testing and validation.

Beven and Kirkby (1979) developed a hydrologic model, TOPMODEL, that uses a topographic index to simulate hydrologic processes in hilly basins in humid temperate climates. They conceived the idea that the dynamics of land areas that become saturated during storm events (variable source areas) are largely determined by topography. Runoff usually originates from saturated soil (source) areas near stream channels, and these areas expand and contract during rain events. This expansion and contraction reflects the state of wetness of the watershed and enables the delivery of high volumes of surface runoff when these source areas comprise a significant proportion of a watershed. Overland flow generated from source areas contributes to the storm hydrograph, often displaying a quick rise in flow rate during a rain event. Flow rate decays to base flow after the event. The modeling of source area dynamics requires an understanding of three-dimensional water flow over the land surface. Basin convergence or divergence determines how much water from upslope areas arrives at a given point. The rate of water flow from this point is determined by slope and soil transmissivity (subsurface flow rate at a unit gradient).

The topographic index used by Beven and Kirkby for a particular location is given by the upslope area (a), which converges on a unit contour width divided by the tangent of the local slope angle (β), all expressed as a natural logarithm: $\ln(a/\tan \beta)$. Locations with similar index values have similar soil wetness characteristics. Places that drain large upslope areas and have low local slope (e.g., valleys) have high index values and tend to be wet. Other places, like ridge tops, have low index values and tend to be dry due to small upslope areas with divergent slopes.

Application of TOPMODEL to a watershed requires a histogram of $\ln(a/\tan \beta)$ values for the site, and these are used in a semidistributed fashion on the assumption that locations with similar index values have similar hydrologic response. TOPMODEL has been applied on two instrumented watersheds (Walker Branch and Center Seven Creek) on the Oak Ridge Reservation (ORR) with satisfactory results. The creation of a histogram of topographic indices for a watershed is the goal of digital terrain analysis.

D.1.2 Digital Terrain Analysis Methods

Digital terrain analysis requires the conversion of a contour (vector) map to a grid cell or raster form to permit discrete cell-by-cell analysis of topographic properties such as slope, flow direction, and topographic index. Each cell represents a small map area and is an approximation of a continuous landform. Thus the accuracy of the results increases as the cell size decreases. The major advantage of using a matrix of elevations is that computer analysis can be much more readily undertaken with digital values than with line segments from a contour map.

Modern workstations are useful for applying algorithms to digitize a contour map and calculate topographic properties from the digital elevation model (DEM). A view of the land surface can be created by drawing continuous lines through the discrete elevation values of the DEM. Grid cells are typically square and range in size from 3 to 70 m on a side. At the low end of this range,

continuous surfaces are closely approximated. Nevertheless, discrete representation alone does not provide the interpolation of water flow direction needed for hydrologic studies. Specific digital terrain analysis algorithms are required to extract topographic properties from DEM-derived matrices.

A DEM must retain the original runoff pathways evident on the contour map. Thus, all stream beds and intermittent flow paths must be characterized without unnatural dams or depressions. This requirement is not trivial, particularly in landscapes with stream channels smaller than the size of a grid cell. Once this runoff constraint is satisfied, a reliable determination of the derived properties of slope, upslope area, and flow direction can be made. Lines of grid cells with the same elevation (contours) can be viewed as equipotential lines across which flow occurs perpendicularly downgradient.

Current methods for producing digital elevation models use either linear, spline, or minimum curvature algorithms to interpolate values between irregularly spaced contours (Schwartz et al. 1995). The best methods use specifically defined input locations for streambeds and ridge lines and apply runoff constraints such that (1) flow cannot cross a ridge and (2) flow along a stream must occur without artificial dams or depressions. Such dams and depressions (or pools) are undesirable because they introduce artificial surface water storages. This problem is determined in part by cell size resolution and in part by the interpolation method used in the digitizing step. The interpolation method is not as important as the cell size resolution in determining the extent of this problem. A grid size of 3 m is used for development of the DEM for the SWSA 6 site. The quality of the DEM is checked by generating views to confirm the absence of spurious features and using flow matrices to confirm the positions of flow pathways.

Terrain analysis of the SWSA 6 site requires that the site contours be modified to account for the landscape grading needed for construction of caps on the three aboveground waste sites (Tumulus I and II and the Interim Waste Management Facility). The site plan is altered to include the proposed contours, and new contours are drawn to show the expected site grading. This map (Fig. 4.9) is modified where necessary to reflect drainage patterns.

The DEM for the SWSA 6 site is prepared with the CGRID program (Timmins, Huston, and Clapp 1989), which uses linear interpolation between contours and the ridge and channel lines to develop the digital model. Contours at a 2-ft interval and stream and ridge lines are digitized and used to produce a DEM with 0.1-ft vertical resolution and 10-ft horizontal resolution. The nominal 0.1-ft vertical resolution permits differencing for calculation of slopes. The raw DEM is viewed as a three-dimensional model to remove spurious features, and a check-and-replace algorithm in the CGRID program is used to smooth any irregularities. DEM values for which the absolute value of the elevation minus the mean of the surrounding eight values exceeds a tolerance of 0.2 feet are replaced by the mean. About 3% of the values were edited to obtain the final DEM for the SWSA 6 site.

D.1.3 Terrain Analysis of the SWSA 6 Site

An overlay mask is made to provide appropriate boundaries for the SWSA 6 site. The DEM is clipped to these boundaries and used in calculation of topographic indices from upslope area and slope angle properties of the landscape (Schwartz et al. 1995). The frequency distribution (histogram) of topographic index values is used in TOPMODEL for hydrologic modeling. This histogram shows a characteristic lognormal distribution (Fig. 4.10), indicating large areas with low soil wetness and relatively small areas of wetlands and stream channels. This map of topographic indices is used to assign spatially varying recharge values for groundwater modeling (Fig. 4.11).

D.2 UNIFIED TRANSPORT MODEL DETERMINATION OF SWSA 6 WATER BUDGETS

UTM is a water budget simulator that includes algorithms for interception storage, interception evaporation, infiltration, soil water drainage, soil evaporation, transpiration, and subsurface lateral flow (Fig. 3.6). The code is documented in Patterson et al. (1974) and Huff et al. (1977b), and hydrologic simulations have been tested with data from Walker Branch watershed on ORR (Huff et al. 1977a; Luxmoore and Huff 1989). The code operates with 15-min time steps when raining, otherwise hourly, to generate the water dynamics for up to seven soil-plant segments of a watershed. The lateral flow and drainage from each land segment can be assembled to generate stream-flow simulations, but this option is not required for site water budget calculations.

The meteorological data needed for simulation with UTM are hourly precipitation, daily maximum and minimum air temperature, daily total solar radiation, daily average dew point temperature, and daily average wind speed (Fig. 4.13). Records from the Walker Branch watershed and from the meteorological station at Oak Ridge townsite have been compiled into the model format for application at sites in the vicinity of Oak Ridge National Laboratory. An evaluation of the data for the 1968–1989 period showed that 1971 was a year with average precipitation (1372 mm/year), 1968 was a dry year (933 mm/year), and 1973 was a wet year (1895 mm/year). These three contrasting sets of data were selected for simulation to provide a range of hydrologic transport regimes. The data for the average year were used in the SWSA 6 assessment simulations. Model applications are initialized by repeating simulations with the same weather data for a second year, using soil water results at the end of the first year to initiate the repeated simulation. The change in annual soil water storage is close to zero with this initialization procedure, and thus there is little or no influence of initial conditions on the simulated water budgets for the various stages of SWSA 6 operations.

It is assumed that all chemicals leached from waste sites are transported in drainage water, even if the volume is small. Because of the dilution effect of drainage water mixing with leachate, chemical solubility is not a limitation. During pad loading operations for tumulus units, water budget

simulations are not used for estimation of seepage into concrete vaults because it was assumed that vaults are exposed to precipitation before completion of the compacted soil cap. Monthly precipitation values are used as seepage input for simulations with the SOURCE models (Table 4.8 and Table 4.9).

Water budget simulations are conducted for a unit with a compacted soil cap. The soil hydraulic properties used in these simulations (Table D.1) involved 60 cm of top soil in two layers over three layers of compacted soil. The saturated conductivity of the top soil layers was 3.5×10^{-5} m/s. The three compacted soil layers of the cap (60–80, 80–100, and 100–120 cm) were given a saturated hydraulic conductivity of 1.2, or 2.3×10^{-7} m/s (Table D.1). These three layers were effective in reducing drainage and enhancing lateral flow in the second layer. The soils of the SWSA 6 site have been described and classified by Davis et al. (1987), and many of the profiles are shallow, having been formed on weathered interbedded shale and limestone (saprolite). The hydraulic properties selected to represent these soils are derived from published sources reporting investigations conducted for the same geologic formation. The hydraulic properties of the subsoil deeper than 120 cm (Table D.1) are based on data from Rothschild et al. (1984). The water retention data are obtained from Luxmoore (1982) and Davis et al. (1987). Water budget simulations for the belowground waste zones during the active use period were conducted with the soil hydraulic properties shown in Table D.2, with one exception. The saturated hydraulic conductivity for the 60- to 120-cm soil layer is 1.5×10^{-5} m/s in this case. These simulations are conducted for both grassy and gravelly surface conditions to represent the dominant surface types at SWSA 6 (Table 4.15).

At the next stage in the tumulus water budget simulations, soil macropores are introduced into the cap to represent the influence of cracks and channels that develop through the matrix material with time. These preferential flow paths progressively negate the hydrologic impedance of the compacted soil matrix. The same water retention data are used for simulation of cap failure and for the post-institutional-control period with forest vegetation (Table D.2). Total porosity varied from 0.390 to 0.476 m³/m³ in the 5-m profile. The water retention data are obtained from Luxmoore (1982) and Davis et al. (1987). Saturated conductivity data for the natural soil layers above a 3-m depth (i.e., excluding cap) are obtained from Rothschild et al. (1984). The saturated hydraulic conductivity values for the bottom two layers are decreased from the values given by Rothschild et al. (1984) in an approximately exponential decrease such that the conductivity of the lowest layer is 5.8×10^{-9} m/s (0.5 mm/d). Simulations using these hydraulic properties (Table D.2) gave lateral flow and drainage results that agreed with the hydrologic framework outlined in Sect. 3.1.5. This framework projects that about 10% or more of the subsurface flow will become recharge to aquifers, with the remainder generating lateral subsurface flow through a storm-flow zone located in the upper 2–3 m of the soil profile. In agreement with recent analyses (Sect. 3.1.5.2), our simulations allow about 30% of subsurface flow to become recharge. Saturated hydraulic conductivity measurements show a decrease with depth, and values are about three orders of magnitude smaller at a 4- to 5-m depth than at the surface (Moore 1988).

Table D.1. Volumetric soil water content (m^3/m^3) at selected matric pressures and the saturated hydraulic conductivity for a compacted soil cap with grass vegetation and for soil layers below tumulus pads

Matric pressure (-kPa)	Soil layer (cm)							
	Top soil		Compacted soil cap			Subsoil		
	0-30	30-60	60-80	80-100	100-120	120-130	130-140	140-160
0	0.476	0.476	0.390	0.390	0.390	0.455	0.455	0.455
1	0.472	0.472	0.384	0.384	0.384	0.432	0.432	0.432
2	0.468	0.468	0.378	0.378	0.378	0.418	0.418	0.418
3	0.466	0.466	0.373	0.373	0.373	0.410	0.410	0.410
4	0.463	0.463	0.369	0.369	0.369	0.402	0.402	0.402
5	0.462	0.462	0.366	0.366	0.366	0.396	0.396	0.396
10	0.452	0.452	0.352	0.352	0.352	0.372	0.372	0.372
15	0.439	0.439	0.342	0.342	0.342	0.354	0.354	0.354
18	0.430	0.430	0.336	0.336	0.336	0.347	0.347	0.347
33	0.370	0.370	0.330	0.330	0.330	0.330	0.330	0.330
100	0.355	0.355	0.305	0.305	0.305	0.305	0.305	0.305
200	0.340	0.340	0.295	0.295	0.295	0.295	0.295	0.295
400	0.310	0.310	0.275	0.275	0.275	0.275	0.275	0.275
800	0.250	0.250	0.250	0.250	0.250	0.250	0.250	0.250
1500	0.140	0.140	0.230	0.230	0.230	0.230	0.230	0.230
Saturated hydraulic conductivity (m/s)	3.5×10^{-5}	3.5×10^{-5}	1.2×10^{-7}	2.3×10^{-7}	1.2×10^{-7}	9.8×10^{-6}	9.8×10^{-6}	2.3×10^{-7}

Spatial variability of hydraulic properties in soil profiles similar to the SWSA 6 site have been determined for the surface soil (Wilson and Luxmoore 1988) and for subsoil (Luxmoore, Spalding, and Munro 1981; Wilson, Alfonsi, and Jardine 1989). These three studies show lognormal frequency distributions for soil-water flow rates, with little or no spatial correlation between measurement sites. Spatial variability of soil properties at the SWSA 6 site is not included in the present analysis.

D.3 ALGORITHMS IN THE PADSIM AND HOLSIM MODELS

The PADSIM/HOLSIM models are developed to simulate vertical and lateral chemical transport through the vadose zone using annual time-step calculations. The choice of annual time steps to represent the dynamics of storm event-driven solute transport was based on the need for

Table D.2. Volumetric soil water content (m^3/m^3) at selected matric pressures and the saturated hydraulic conductivity of a failed compacted soil cap and soil layers with grass or forest vegetation

Matric pressure (-kPa)	Soil layer (cm)							
	Top soil	Cap	Subsoil					
	0-60	60-120	120-150	150-200	200-250	250-300	300-400	400-500
0	0.476	0.419	0.455	0.456	0.390	0.390	0.390	0.390
1	0.472	0.410	0.432	0.448	0.384	0.384	0.384	0.384
2	0.468	0.403	0.418	0.440	0.378	0.378	0.378	0.378
3	0.466	0.397	0.410	0.433	0.373	0.373	0.373	0.373
4	0.463	0.392	0.402	0.429	0.369	0.369	0.369	0.369
5	0.462	0.388	0.369	0.425	0.366	0.366	0.366	0.366
10	0.452	0.369	0.372	0.406	0.352	0.352	0.352	0.352
15	0.439	0.352	0.354	0.389	0.342	0.342	0.342	0.342
18	0.430	0.346	0.347	0.380	0.336	0.336	0.336	0.336
33	0.370	0.310	0.330	0.350	0.330	0.330	0.330	0.330
100	0.355	0.300	0.305	0.325	0.305	0.305	0.305	0.305
200	0.340	0.285	0.295	0.315	0.295	0.295	0.295	0.295
400	0.310	0.270	0.275	0.295	0.275	0.275	0.275	0.275
800	0.250	0.250	0.250	0.270	0.250	0.250	0.250	0.250
1500	0.140	0.230	0.230	0.250	0.230	0.230	0.230	0.230
Saturated hydraulic conductivity, m/s	3.5×10^{-5}	2.3×10^{-9a}	9.8×10^{-6}	2.3×10^{-6}	5.8×10^{-7}	5.8×10^{-7}	3.5×10^{-8}	5.8×10^{-9}

^a Conductivity was enhanced by a macropore algorithm.

assessment simulations for many thousands of years. PADSIM is designed for simulation of aboveground tumulus structures, and calculations are made on a per pad basis. HOLSIM is applied to belowground waste units and accounts for units of differing ages resulting from sequential filling during the active use period. PADSIM and HOLSIM have algorithms for (1) preferential flow as a function of K_d for soil within 3.5 m of the surface, (2) an annual increment in reactive soil path length as a function of K_d , (3) half-life calculations of nuclide decay, (4) chemical partitioning between solution-phase concentration and adsorption on soil surfaces, (5) matrix diffusion of nuclide into matrix during bypass flow, and (6) mass balance calculations. An outline of these algorithms is given below and includes some FORTRAN coding statements. FORTRAN variables are given in capital letters; for example, CKD represents K_d .

D.3.1 Preferential Flow Algorithm

Preferential flow from the shallow subsurface is not connected to the water table below a depth of 3.5 m in the vadose zone. However, preferential flow has been shown to be a significant component of lateral flow through the shallow subsurface above a depth of 3 m (Wilson et al. 1993). The annual proportion of bypass flow to surface water is made an arbitrary function of K_d such that a smaller proportion of bypass flow results for nuclides with higher K_d .

All nuclides with a K_d (CKD) greater than 10 are allowed to migrate a maximum annual distance (BYLEN) of 10 m. Nuclides with a lower K_d have a linearly greater annual transport distance with decrease in K_d . The following FORTRAN statements calculate BYLEN (m) and determine the proportion of bypass flow, SBYPAS, as a function of STLEN, an input value for the distance (m) of the lateral flow path from each waste zone to the nearest surface drainage channel. The proportion of bypass flow to the water table (RBYPAS) is a function of the distance between the bottom of the waste units and the water table (GWDIS).

```
IF(CKD.GT.10.) THEN
  BYLEN = 10.0 - (0.003*CKD)
ELSE
  BYLEN = 270.0 - (26. * CKD)
ENDIF
RBYPAS = 0.0
SBYPAS = 0.0
IF(GWDIS.LT.BYLEN) RBYPAS = 1.0 - (GWDIS/BYLEN)
IF(GWDIS.GT.3.5) RBYPAS = 0.0
IF(STLEN.LT.BYLEN) SBYPAS = 1.0 - (STLEN/BYLEN)
```

Nuclides with a K_d of zero have a BYLEN value of 270 m; thus, with a STLEN value of 47 m for the site, the value for SBYPAS is 0.826. SBYPAS decreases to zero for a K_d of 9 or more (Table 4.16). This simplistic treatment of preferential flow could be refined in continuing work.

D.3.2 Annual Increment in Soil Path Length

An annual increment in the soil path length (ANINC) exposed to nuclide is determined as a proportion of the maximum annual transport distance (BYLEN) obtained in the preferential flow calculations. An arbitrary fraction (FPAINC) of 0.05 is applied to BYLEN to obtain ANINC values that decrease from 4 m for nuclides with a zero K_d to 0.75 m for nuclides with a $K_d \geq 100$. The FORTRAN statement is

```
ANINC = BYLEN * FPAINC .
```

These calculations result in small annual increments in the soil matrix path length that add to the maximum annual transport length given by BYLEN. These increments increase the soil volume interacting with nuclides. Because the maximum recharge path length of 4.7 m is much shorter than the maximum annual transport distance for many adsorbed nuclides ($K_d \leq 1000$), the whole of the soil volume in the recharge path interacts with these nuclides in all years. However, 75 years are required for transport of cesium (with a K_d of 3000) through the 4.7 m of the vadose zone to the water table (Table 4.17). Adsorbed nuclides in the lateral subsurface flow path take decades to millenia to reach a stream channel. For example, cesium takes 921 years to be transported 47 m (Table 4.17). Unretarded species (those with a K_d of 0) interacted with the whole lateral flow path in all years.

D.3.3 Half-Life Calculations

Nuclide decay is calculated each year for the nuclide mass in the vertical and lateral subsurface flow paths. Nuclide decay for the adsorbed phase in the recharge path (RSOLID) reduces this phase according to the FORTRAN statement

$$RSOLID = RSOLID * DEXP[-(HARATE/HALIFE)] ,$$

where HARATE is $\log 2.0$ in double precision and HALIFE is the nuclide half-life in years. A similar statement is used for solution-phase calculations.

D.3.4 Chemical Adsorption

The relationship between solution-phase and solid-phase partitioning of nuclides is calculated on two occasions in PADSIM. The first is for incorporation of the annual addition of nuclide from SOURCE1 into the soil column; the second is the adjustment in nuclide partitioning after nuclide export from the soil column. Calculations of transport through the lateral flow path to surface water and through the recharge path to groundwater are conducted independently for independent soil volumes.

The key equations for adsorption calculations are shown using the recharge path (FORTRAN variables beginning with the letter R) as an example. The determination of the new solution concentration (RSOLU) in the recharge path after the annual addition of nuclide and water from SOURCE1 calculations involves the following statements.

1. The sum of adsorbed chemical (RSOLID, μg) and chemical in solution (RLIQUID, μg) in the soil column from the previous time step plus the new nuclide addition [RLCH (I), μg] gives the total nuclide content of the soil column (RTCHEM, μg). New values [identified by "?" in Eq. (D.1)] for RSOLID and RLIQUID need to be determined for the revised total nuclide in the soil column.

$$\begin{array}{ccc} \text{RSOLID} & + & \text{RLIQUID} = \text{RTCHEM} \\ ? & & ? \quad (\text{known}) \end{array} \quad (\text{D.1})$$

2. The water content of the recharge path (RWATER, mL) is also adjusted from field capacity (RFDCAP, mL) by the addition of the vertical water flux (RFLOWV, mL) from SOURCE1 calculations. The current soil mass (RSMAS, g) in the flow path is given by the product of the current soil volume (RSVOL, mL) and the mean bulk density (BD, g/mL).
3. The K_d (CKD, mL/g) relationship has the form

$$\text{CKD} = \frac{\text{RSOLID}/\text{RSMAS}}{\text{RLIQUID}/\text{RWATER}} \quad (\text{D.2})$$

4. The determination of solution concentration (RSOLU, $\mu\text{g/mL}$) at the new equilibrium is obtained in the following steps:

- (a) The left-hand side of Eq. (D.2) is multiplied by RSVOL/RSVOL to give

$$\text{CKD} = \frac{\text{RSOLID}/\text{BD}}{\text{RLIQUID}/\text{RWPOR}} \quad (\text{D.3})$$

where RWPOR is the water-filled matrix porosity given by RWATER/RSVOL in units of mL/mL.

- (b) Substituting Eq. (D.1) in Eq. (D.3) to define RSOLID in terms of RLIQUID and rearranging to obtain an expression for RLIQUID:

$$(\text{RTCHEM} - \text{RLIQUID})/\text{BD} = \text{CKD} * \text{RLIQUID}/\text{RWPOR}$$

$$\text{RTCHEM} - \text{RLIQUID} = \text{CKD} * \text{BD} * \text{RLIQUID}/\text{RWPOR}$$

$$\text{RLIQUID}[1 + (\text{CKD} * \text{BD}/\text{RWPOR})] = \text{RTCHEM}$$

$$\text{RLIQUID} = \text{RTCHEM}/[1 + (\text{CKD} * \text{BD} * \text{RWPOR})] \quad (\text{D.4})$$

- (c) Divide RLIQUID by RWATER to get RSOLU from Eq. (D.4):

$$\text{RSOLU} = \text{RTCHEM}/[\text{RWATER} + (\text{CKD} * \text{BD} * \text{RSVOL})]$$

5. The flux of nuclide to groundwater (RFLUX, μg) is given by the product of solution concentration (RSOLU, $\mu\text{g/mL}$) and the recharge volume (RFLOWV, mL).

The second adjustment of solid and liquid chemical partitioning is made after outflow of nuclide in recharge to groundwater. RTCHEM is reduced by the amount of chemical loss (RFLUX) in recharge, and RWPOR is reduced to RFDCAP/RSVOL. Thus, the total nuclide mass remaining in the soil path (RSOIL) is given by

$$\text{RSOIL} = \text{RTCHEM} - \text{RFLUX} .$$

The proportion of the total nuclide associated with the solution phase (RLIQUID) is given by

$$\text{RLIQUID} = \text{RSOIL} / [1 + (\text{CKD} * \text{BD} * \text{RFDCAP} / \text{RSVOL})] ,$$

and the adsorbed phase is given by

$$\text{RSOLID} = \text{RSOIL} - \text{RLIQUID} .$$

These revised values of RLIQUID and RSOLID are used in calculations at the next time step.

D.3.5 Matrix Diffusion During Bypass Flow

Research has shown that solutes moving through preferential flow paths can transfer into the matrix. We have incorporated this effect into PADSIM and HOLSIM such that the proportion of solute transferred to the matrix (RDM, SDM) increases with increase in pathlength (GWDIS, STLEN) up to a limit of 95%, as follows:

$$\text{RDM} = 0.0035 * \text{GWDIS}$$

$$\text{IF}(\text{RDM} > 0.95) \text{ RDM} = 0.95$$

$$\text{SDM} = 0.0035 * \text{STLEN}$$

$$\text{IF}(\text{SDM} > 0.95) \text{ SDM} = 0.95$$

D.3.6 Mass Balance Test

PADSIM and HOLSIM conduct a calculation to determine the difference between the sum of inflow and outflow of nuclide at a particular time step and the change in nuclide content within the

soil column for the time step. An error message is printed if this difference exceeds machine round-off error. No error messages were obtained for the simulation results presented in this PA

D.4 REFERENCES

- Beven, K. J., and M. J. Kirkby 1979. "A Physically-Based Variable Contributing Area Model of Basin Hydrology," *Hydrological Sciences Bulletin* 24, 43-69.
- Davis, E. C., et al. 1987. *Summary of Environmental Characterization Activities at the Oak Ridge National Laboratory Solid Waste Storage Area Six, FY 1986 through 1987*, ORNL/RAP/LTR-87/68, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Huff, D. D., et al. 1977a. "The Application of Analytic and Mechanistic Hydrologic Models to the Study of Walker Branch Watershed," pp. 741-63 in *Watershed Research in Eastern North America*, ed. D. L. Correll, Chesapeake Bay Center for Environmental Studies, Smithsonian Institution, Edgewater, Md.
- Huff, D. D. et al. 1977b. *TEHM: A Terrestrial Ecosystem Hydrology Model*, ORNL/NSF/EATC-27, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Luxmoore, R. J. 1982. *Physical Characteristics of Soils of the Southern Region*, ORNL-5868, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Luxmoore, R. J., and D. D. Huff 1989. "Water," pp. 164-96 in *Analysis of Biogeochemical Cycling Processes in Walker Branch Watershed*, ed. D. W. Johnson and R. I. Van Hook, Springer-Verlag, New York.
- Luxmoore, R. J., B. P. Spalding, and I. M. Munro 1981. "Areal Variation and Chemical Modification of Weathered Shale Infiltration Characteristics," *Soil Science Society of America Journal* 45, 687-91.
- Moore, G. K. 1988. *Concepts of Groundwater Occurrence and Flow near Oak Ridge National Laboratory, Tennessee*, ORNL/TM-10969, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Patterson, M. R., et al. 1974. *A User's Manual for the Fortran IV Version of the Wisconsin Hydrologic Transport Model*, ORNL/NSF/EATC-7, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Quinn, P. et al. 1991. "The Prediction of Hillslope Flow Paths for Distributed Hydrological Modelling Using Digital Terrain Models," *Hydrological Processes and Water Management* 5, 3-30.
- Rothschild, E. R., et al. 1984. *Characterization of Soils at Proposed Solid Waste Storage Area (SWSA) 7*, ORNL/TM-9326, Oak Ridge National Laboratory, Oak Ridge, Tenn.

- Schwartz, P. M., et al. 1995. *TERRAIN: A Computer Program to Process Digital Elevation Models for Modeling Surface Flow*, ORNL/TM-13071, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Timmins, S. P., M. A. Huston, and R. B. Clapp 1989. "Generation of Digital Elevation Models from Contours and Comparison of Topographic Indices," *Eos, Transactions of the American Geophysical Union* **70**, 1094.
- Wilson, G. V., J. M. Alfonsi, and P. M. Jardine 1989. "Spatial Variability of Saturated Hydraulic Conductivity of the Subsoil of Two Forested Watersheds," *Soil Science Society of America Journal* **53**, 679-85.
- Wilson, G. V., and R. J. Luxmoore 1988. "Infiltration, Macroporosity, and Mesoporosity Distributions on Two Forested Watersheds," *Soil Science Society of America Journal* **52**, 329-35.
- Wilson, G. V., et al. 1993. "Field-Scale Transport from a Buried Line Source in Variably Saturated Soil," *Journal of Hydrology* **145**, 83-109.

Appendix E

GROUNDWATER FLOW AND CONTAMINANT TRANSPORT MODELING FOR THE SWSA 6 PERFORMANCE ASSESSMENT

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Appendix E

GROUNDWATER FLOW AND CONTAMINANT TRANSPORT MODELING FOR THE SWSA 6 PERFORMANCE ASSESSMENT

This appendix presents a brief summary of the theory of porous-medium contaminant transport, followed by the development and application of a groundwater model for Solid Waste Storage Area 6 (SWSA 6). SWSA 6 is the only low-level radioactive waste (LLW) disposal facility currently operating on the Oak Ridge Reservation (ORR) and is the subject of this performance assessment (PA).

E.1 POROUS-MEDIUM CONTAMINANT TRANSPORT THEORY

Transport equations are founded on the principle of the conservation of mass: the net rate of increase of mass in a control volume is equal to net mass flux into the volume plus the increase of mass within the volume. This statement of continuity is expressed in the equation

$$\frac{\partial m}{\partial t} + \nabla \cdot \mathbf{J} = S^+ \quad , \quad (\text{E.1})$$

where

- m $\equiv$ bulk concentration (units of mass/length³),
- t $\equiv$ time (units of time),
- $\mathbf{J}$ $\equiv$ the mass flux vector, representing the mass transport across a unit surface per unit time, normal to the surface (units of mass/length²/time),
- S^+ $\equiv$ strength of mass production within the control volume (units of mass/length³/time),

and $\nabla \cdot \mathbf{J}$ is the divergence of the flux vector. The bulk concentration m of the constituent includes all of the mass in the volume, be it sorbed or in solution. Assuming local equilibrium and a linear sorption isotherm, the sorbed concentration C_s can be expressed as a function of the aqueous concentration C_{aq} :

$$C_s = K_d C_{aq} \quad , \quad (\text{E.2})$$

where K_d is the distribution coefficient for the constituent. The bulk concentration is the sum of the aqueous and sorbed concentrations:

$$m = n C_{aq} + \rho_b C_s , \quad (\text{E.3})$$

or

$$m = (n + \rho_b K_d) C_{aq} , \quad (\text{E.4})$$

where

- ρ_b $\equiv$ soil density (units of mass/length³),
 n $\equiv$ total porosity (volume of voids/total volume).

The mass flux vector J includes both advective and dispersive components as described above:

$$J = q C - n D \cdot \nabla C , \quad (\text{E.5})$$

where

- q $\equiv$ Darcy flux, or specific discharge vector (units of length³/time/length²),
 D $\equiv$ dispersion tensor (units of length²/time).

The right-hand side of Eq. (E.1) contains the source term, S^+ . The only process considered in the PA which affects this value is decay as a first-order function of mass, so that

$$S^+ = -\lambda m . \quad (\text{E.6})$$

Substituting these expressions into Eq. (E.1) gives

$$\frac{\partial}{\partial t} \left[(n + \rho_b K_d) C \right] + \nabla \cdot \left[q C - n D \cdot \nabla C \right] = -\lambda \left[(n + \rho_b K_d) C \right] . \quad (\text{E.7})$$

To simplify, let us assume steady flow, with no volumetric sources or sinks. In this case,

$$\nabla \cdot q = 0 . \quad (\text{E.8})$$

By the chain rule,

$$\nabla \cdot (q C) = C \nabla \cdot q + q \cdot \nabla C , \quad (\text{E.9})$$

the second term of which disappears by Eq. (E.8). Assuming also a constant fluid density and porosity, Eq. (E.7) can be written

$$(n + \rho_b K_d) \frac{\partial C}{\partial t} + q \cdot \nabla C - n \nabla \cdot (D \cdot \nabla C) = -\lambda C (n + \rho_b K_d) \quad (\text{E.10})$$

The definition of retardation can simplify this equation further:

$$n R \frac{\partial C}{\partial t} + q \cdot \nabla C - n \nabla \cdot (D \cdot \nabla C) = -\lambda C n R \quad (\text{E.11})$$

Dividing through by the porosity n gives the general form

$$R \frac{\partial C}{\partial t} - \nabla \cdot (D \cdot \nabla C) + v \cdot \nabla C + \lambda R C = 0 \quad (\text{E.12})$$

This is the principal governing equation for the following exercise in modeling advection-dispersion.

One of the central issues in modeling contaminant transport is the derivation of the flow field. If the hydraulic conductivity tensor $\mathbf{K}$ is isotropic, groundwater flows in the direction of the largest gradient in the hydraulic head surface. Even though it is a common assumption that $\mathbf{K}$ is isotropic, in the vicinity of Oak Ridge it is strongly anisotropic, and groundwater flow does not simply follow the largest gradient, but rather follows the strike of the tilted geologic strata. This behavior still falls within the domain of classical porous medium physics and can be modeled by application of the head equation (Bear 1972),

$$\frac{\partial}{\partial x} \left(K_x \frac{\partial h}{\partial x} \right) + \frac{\partial}{\partial y} \left(K_y \frac{\partial h}{\partial y} \right) + S_s \frac{\partial h}{\partial t} = 0 \quad (\text{E.13})$$

together with appropriate initial and boundary conditions. Here K_x and K_y are x - and y - components of the aquifer hydraulic conductivity tensor, h is the hydraulic head, and S_s is the specific storage. The solution of this equation may be constrained in several ways, principally by adjusting boundary conditions and flow in and out of the system. In the best case, the computed head replicates the measured head at measurement locations.

E.2 GROUNDWATER MODELING FOR SWSA 6

This section discusses the development and application of the computer groundwater model used to evaluate contaminant transport from SWSA 6 disposal units. The model is an application of the U.S. Geological Survey's (USGS's) Method of Characteristics (MOC) computer program, with specialized material properties and boundary conditions imposed in order to represent the SWSA 6

site. The purpose of the groundwater model is to transport contaminated recharge to various points of exposure.

Some data for the SWSA 6 groundwater flow and contaminant transport model are provided by other models used in the PA. The principal link to the other models is via surface recharge, which defines volumes of water and associated contaminant concentrations introduced to groundwater over its entire modeled area. This recharge-induced flow is routed through a discretized array of cells which represent the subsurface as a porous medium. After traveling through and interacting with subsurface materials, this water eventually returns to the surface water system as discharge to creeks.

Concentrations of radionuclides are monitored at specific locations in the groundwater (hypothetical water supply wells) and in the surface water system in order to evaluate the dose to a hypothetical future individual who would use these waters for drinking.

E.2.1 Conceptual Model

SWSA 6 is situated on the southern slopes of a midvalley mound in western Melton Valley, straddling the contact between the Maryville and Nolichucky formations (Fig. E.1). Near the surface, the weathered regolith is fairly uniform in consistency, so the model does not distinguish between the two source lithologies. This upper layer of the subsurface dominates the local subsurface flow regime, and is modeled as a single layer in a two-dimensional (2-D) groundwater flow and transport model. Below this lies more massive bedrock, with a significantly lower conductivity, and which is assumed to be impermeable, making the "floor" of the model. The model area is dissected by small, intermittent streams that serve to drain surface recharge and may be outlets for deeper groundwater as well, so that no groundwater is expected to cross these tributary drainages. These creeks run in a southerly direction to join White Oak Creek (WOC) or White Oak Lake (WOL). Ultimately, all surface waters drain to WOL, which is impounded by White Oak Dam (WOD). The dam is therefore the outlet of all surface waters that encounter wastes in SWSA 6, as shown in Fig. E.2. For the purposes of this PA, it is assumed that all groundwaters in the WOD watershed also drain to creeks and subsequently to WOD. This assumption is conservative because no contaminants are allowed to exit the model by any other pathway: WOD integrates all contaminant transport.

Major features of the groundwater conceptual model include a steady-state flow field, diffuse recharge (with that recharge through waste disposal units being contaminated), chemical retardation, and discharge of groundwater to streams and surface water drains. Within an annual time step, all groundwater discharges to streams or drains are combined with the discharge from the shallow subsurface transport model.

Fig. E.1. Generalized geologic map of SWSA 6.

SWSA 6 within White Oak Creek Watershed

Fig. E.2. Location of SWSA 6 in the White Oak Creek watershed.

E.2.2 Computer Model

Based on conceptualization of SWSA 6 presented in the previous section, it was determined that groundwater flow and the associated contaminant transport within SWSA 6 could be adequately simulated using an existing 2-D numerical model.

E.2.2.1 The USGS MOC Computer Program

The computer model used to analyze groundwater flow and associated contaminant transport within SWSA 6 was built using MOC, a computer program developed for USGS (Konikow and Bredehoeft 1988). The program may be used to perform 2-D analyses of both steady-state and transient-flow simulations, as well as associated contaminant transport calculations for fully saturated porous media. The "flow" portion of the program solves for hydraulic head (the state variable) based on a finite-difference approximation to the governing equation (E.13). It assumes an invariant water density and viscosity. It can effectively model a wide range of physical geometries including aquifers with both spatially variable hydraulic conductivities and volumetric recharge rates. Additional features allow for the simulation of manmade features such as injection wells. The "transport" portion of the program uses the method of characteristics solve the governing equation for solute transport (E.12). It incorporates the mechanisms of advection, dispersion, linear reversible adsorption, and radioactive decay.

Although there are numerous programs available for solving saturated flow and mass transport problems, MOC was chosen for the analysis for a number of reasons. Features that made it a desirable choice included the relative simplicity and ease of operation of the code, the ability to specify spatially dependent volumetric recharge and associated contaminant concentrations, and the familiarity of the user with the code. The USGS MOC code is written in FORTRAN and has undergone significant modifications as detailed in later sections. All groundwater model runs were conducted on a Hewlett-Packard Model 9000 Series 730 UNIX workstation running HP-UX version 10.00.

E.2.2.2 Groundwater Flow Model

Estimation of contaminant transport phenomena requires the foundation of a calibrated fluid-flow computer model. This section describes the development of a porous medium-flow model for the SWSA 6 site.

E.2.2.2.1 Spatial Discretization

A rectangular region of Melton Valley measuring 2500 by 2750 ft (an area of about 64 ha, or 158 acres, identified in the rectangle drawn in Fig. E.2) surrounding the SWSA 6 site was selected

as the modeling domain based on the location of a number of natural features that served as logical boundaries for a groundwater model. These boundaries include WOL to the south, the creek bordering SWSA 6 to the east, and watershed divides to the north and west. This area was discretized into a regularly spaced, finite-difference grid consisting of 50 cells in the *X* (easting, strike parallel) direction and 55 cells in the *Y* (northing, strike perpendicular) direction, for a total of 2750 cells, each 50 ft square. A plan view of the model grid in relation to selected natural features and the SWSA 6 boundary is shown in Fig. E.3. Distances are given in feet rather than meters because the administrative grid (the origin of the easting and northing coordinate system) is defined in feet, and all references to locations of wells and other geographic elements are given in this system. About one-third of the cell in the rectangular boundary are not part of the modeling domain proper because they lie outside the natural boundaries mentioned above. These inactive cells are given the "no-flow" boundary condition to effectively remove them from the model.

Waste disposal units of various types are distributed throughout the modeling domain. From the perspective of the groundwater model, the type of unit is irrelevant — contamination is introduced simply as a concentration of each radionuclide in a given volume of recharge water added to each cell in the domain. Cells that do not contain disposal units provide clean recharge, but those with wastes must be identified. For the purposes of the PA, only those wastes emplaced after September 26, 1988, are assumed to exist. These disposal units were grouped into zones of like and generally proximal units, and each zone was then assigned to one or more cells. This discretization of sources is shown in Fig. E.4.

E.2.2.2.2 Hydrogeologic and Modeling Assumptions

The computer model developed for groundwater flow and transport calculations at SWSA 6 is of a degree of sophistication consistent with the data used to drive it. As described in the following section, most of the essential material properties defining the porous medium are not sufficiently characterized to justify spatial variation, and are therefore given constant values throughout the domain. Among these constant-valued properties are hydraulic conductivity and transmissivity, which results in a constant aquifer thickness. The bulk of contaminant transport is assumed to occur in the weathered regolith, which is modeled as a 2-D array of cells of a constant thickness, and with a confined bottom. Water enters the cell array through cell tops as recharge, and exits the model only through discharge to surface waters, modeled as drains. Ultimately, all surface waters from SWSA 6 exit the watershed over WOD. The flow field is steady-state, as governed by the constant recharge flux.

A summary of modeling assumptions follows:

- Groundwater flow and contaminant transport may be modeled as a 2-D phenomenon. This implies that material properties do not change with depth and that contaminants are mixed uniformly over the depth of the aquifer.

Relationship of SWSA 6 Groundwater Flow Model Cells to Waste Sites

Fig. E.3. Discretization of the model grid and location of modeled LLW disposals in SWSA 6.

Discretization of Post-9/88 SWSA 6 Disposal Facility Zones

Fig. E.4. Discretization of waste disposal zones in SWSA 6.

- Groundwater flow may be modeled as a steady-state phenomenon. This implies that material properties and flow velocities do not change over time.
- The fractured, weathered subsurface media may be modeled using an effective porous medium model.
- Darcy's law is valid at all points, in that the groundwater flow can be modeled as a single porous medium; unstressed flow is determined solely by hydraulic gradient, and natural stresses such as recharge and discharge may be described as driving terms that are consistent with Darcy's law.
- The fluid is assumed to be fresh water, with no variation in density, salinity, or temperature throughout the domain.
- Ionic and molecular diffusions do not effectively influence dispersive flux.
- The hydraulic stresses on the aquifer are given solely by recharge and discharge to streams, drains, and WOL.
- The aquifer at SWSA 6 may be modeled as homogeneous and anisotropic with respect to hydraulic conductivity and homogeneous and isotropic with respect to dispersivity.
- The movement of each radionuclide is retarded in the porous medium by equilibrium-controlled sorption or ion exchange; the sorption-concentration may be described by a linear isotherm.
- The radionuclide solutes are independent in the sense that the concentration of one is not coupled to the concentration of another. Thus, the superposition of concentrations is valid.

E.2.2.2.3 Material Properties

Material properties defined for the groundwater flow and transport model include porosity, aquifer thickness, and solid-phase bulk density. Properties involving both the fluid and solid phases are hydraulic conductivity and dispersivity. All of these properties are constant-valued over the model, with values presented in Table E.1. Detailed discussion of the derivation of these values follows.

Table E.1. General SWSA 6 groundwater model input parameters

Porosity	0.05
Longitudinal dispersivity	3.0 ft
Transverse dispersivity	0.3 ft
Hydraulic conductivity, K_x	1.73 ft/d
Anisotropy of conductivity tensor, K_x/K_y	3.0
Aquifer saturated thickness	9.8 ft
Soil bulk density	1.35 g/mL
Specific storage	0.0×10^{-6}

- *Bulk density.* The bulk density of all geologic materials is set at 1.35 g/mL, consistent with the shallow subsurface modeling.
- *Porosity.* A value of 0.05 (5%) effective porosity was used. This value is obtained from aquifer tests performed at the Engineered Test Facility (ETF), located in SWSA 6 (Davis et al. 1984).
- *Dispersivity.* The following fixed values of dispersivity are used: $\alpha_L = 0.9$ m (3.0 ft), and $\alpha_T = 0.09$ m (0.3 ft). These values are consistent with those reported by Lee et al. (1989).

Many of the parameter values used for the simulations were determined by simply using those calibrated at the Bear Creek test site or the ETF. Others were determined by other means, such as adjustment by iterative processes. Those which benefitted from more detailed analyses are described below.

Hydraulic Conductivity

The hydraulic conductivity (K) field used for modeling the SWSA 6 site is uniform and anisotropic. After analyzing the available information concerning estimates of K at a number of wells, no spatial correlation or anisotropy to the field could be justified, and the use of a single value of K is reasonable. This value was the primary variable adjusted in order to calibrate the MOC groundwater model. A discussion of the K dataset and the ensuing geostatistical analysis follows.

An extensive set of estimates of hydraulic conductivity for wells in Melton Valley was obtained from the Oak Ridge Environmental Information System (OREIS) database, and was filtered to include just those wells within 1000 ft (300 m) of the SWSA 6 modeling grid. Since the bulk of groundwater moves in the upper layers of the subsurface, it was necessary to examine the distribution of hydraulic conductivity as a function of depth. Saturated hydraulic conductivity at the SWSA 6 site, as elsewhere in Conasauga terrain, generally decreases with depth, and at any discrete depth the conductivity may vary within two to three orders of magnitude (see Fig. E.5). The K estimates were further filtered to include only those from tests performed at less than 10 m (about 30 ft) below ground surface. The complete list of these "shallow" hydraulic conductivity estimates is given in Table E.2, and constitutes the data set used for geostatistical analysis.

Table E.2 In situ estimates of hydraulic conductivity used in the derivation of groundwater modeling parameters for SWSA 6

Well ID	Easting (ft)	Northing (ft)	Total depth (ft)	K depth (ft)	K (ft/d)	$\log(K)$ (ft/d)
371	25090.0	16393.0	29.8	24.8	0.1608	-0.7938
372	25042.0	16570.0	33.2	28.2	0.1312	-0.8820
373	25208.0	16945.0	33.9	28.9	0.0007	-3.1830
374	25346.0	17462.0	30.9	25.9	0.2789	-0.5546

Table E.2. (continued)

Well ID	Easting (ft)	Northing (ft)	Total depth (ft)	K depth (ft)	K (ft/d)	log(K) (ft/d)
376	23479.0	16584.0	34.7	29.7	2.1981	0.3421
380	23320.0	15977.0	31.3	26.3	1.2139	0.0842
381	24266.0	16248.0	35.4	30.4	0.1706	-0.7680
382	24025.0	15814.0	23.1	18.1	0.0558	-1.2536
383	24895.0	16165.0	33.2	28.2	0.2198	-0.6579
642	24035.6	16581.0	30.0	22.5	0.0787	-1.1038
644	24848.4	16748.9	13.5	12.0	0.0558	-1.2536
645	25274.6	17173.7	25.0	20.0	0.4265	-0.3701
735	25907.1	16707.9	8.9	7.5	0.3609	-0.4426
736	25969.3	16562.7	9.0	6.5	0.0558	-1.2536
737	25992.6	16434.9	6.8	6.0	1.0827	0.0345
738	25498.8	16134.2	8.5	6.0	0.1148	-0.9400
739	23604.0	15628.9	33.0	28.0	0.4921	-0.3079
740	23372.4	15779.0	23.9	19.0	0.0755	-1.1223
741	23352.1	15596.7	22.0	17.0	0.7874	-0.1038
742	23338.6	15304.2	33.0	28.0	4.9212	0.6921
743	23248.6	15098.2	25.7	22.0	0.9514	-0.0216
744	23438.7	15165.2	31.5	28.0	0.4593	-0.3379
747	25064.6	15488.1	22.1	17.5	0.0689	-1.1618
748	25534.6	15891.1	22.2	17.5	0.2592	-0.5864
749	26038.6	16314.9	22.1	17.0	0.3051	-0.5155
833	23842.4	16056.9	31.0	20.1	0.4422	-0.3544
835	23951.8	15767.7	27.5	16.2	0.6888	-0.1619
836	24138.8	15748.2	28.5	17.6	0.7455	-0.1276
838	24934.8	16308.7	22.8	13.3	0.3118	-0.5061
840	25251.7	16928.6	26.9	18.8	1.3464	0.1292
842	25298.4	17216.1	26.8	15.6	0.2500	-0.6021
843	25221.4	17597.1	21.0	11.4	0.2976	-0.5263
845	24988.3	17108.2	41.0	29.6	0.9694	-0.0135
849	24215.2	16781.8	32.9	21.6	0.6208	-0.2071
851	24058.2	16485.0	21.6	12.0	0.3090	-0.5101
852	23911.6	16346.9	25.3	13.4	0.9043	-0.0437
853	24047.5	16695.1	27.7	15.2	0.1709	-0.7672
854	23851.9	16711.9	27.5	16.2	1.6696	0.2226
859	23246.2	16009.4	26.5	15.8	1.5590	0.1929
1062	22371.5	15552.1	30.1	25.1	0.0656	-1.1830
1063	22302.2	15356.1	16.0	11.0	0.0722	-1.1416
1074	25707.5	18128.1	30.0	22.5	0.0045	-2.3473
1076	25858.9	17882.7	20.0	14.4	0.0998	-1.0010
1078	25613.7	17406.9	20.0	13.8	0.0349	-1.4575
1080	25971.4	16854.9	20.0	14.7	0.1310	-0.8828
1082	26347.6	16911.0	15.0	10.0	1.0488	0.0207
1185	26464.8	16343.9	27.5	21.9	0.0672	-1.1727
1188	24267.2	15170.5	27.5	22.0	0.0847	-1.0719
1189	23281.0	15146.8	27.5	19.8	0.4082	-0.3891
1191	23388.0	15373.8	26.2	20.8	0.7115	-0.1478
1192	23358.4	15542.4	18.8	12.8	16.1006	1.2068
1659	23641.2	16756.0	28.7	23.7	0.8858	-0.0527
1660	23665.2	16778.3	31.8	26.8	0.0656	-1.1830
1661	23672.8	16764.9	30.8	25.8	0.1411	-0.8506
1662	23673.3	16749.2	30.8	25.8	0.3609	-0.4426
1663	23665.3	16732.5	30.3	25.3	0.8530	-0.0690
1664	23651.0	16724.1	30.1	25.1	1.1155	0.0475

Table E.2. (continued)

Well ID	Easting (ft)	Northing (ft)	Total depth (ft)	K depth (ft)	K (ft/d)	log(K) (ft/d)
1665	23633.6	16723.2	30.4	25.4	0.5577	-0.2536
1666	23618.6	16729.2	29.8	24.8	0.8858	-0.0527
1667	23610.3	16745.3	30.9	25.9	0.1411	-0.8506
1675	23606.4	16782.9	18.5	13.5	0.0272	-1.5649
1678	23606.7	16769.5	21.8	16.8	0.6562	-0.1830
1679	23621.5	16776.5	19.3	14.3	0.0312	-1.5063
1680	23641.2	16788.0	22.2	17.2	0.1280	-0.8930
1681	23659.2	16797.9	20.2	15.2	0.1706	-0.7680
1682	23620.2	16762.6	21.8	16.8	1.7060	0.2320
1683	23638.0	16773.9	21.6	16.6	0.2165	-0.6645
1684	23655.5	16785.0	21.1	16.1	0.1345	-0.8712
1685	23616.4	16745.6	20.4	15.4	0.0623	-1.2053
1686	23632.1	16756.5	20.3	15.3	0.0259	-1.5864
1687	23651.4	16769.4	20.7	15.7	0.0623	-1.2053
1688	23628.8	16743.2	17.3	12.3	0.0689	-1.1618
1689	23646.4	16753.2	19.8	14.8	0.0820	-1.0861
1690	23662.1	16764.5	20.0	15.0	6.8897	0.8382
1691	23623.7	16732.4	22.9	17.9	0.0591	-1.2287
1692	23641.6	16741.0	20.6	15.6	0.0427	-1.3701
1693	23658.6	16749.1	21.2	16.2	0.0853	-1.0690
1694	23674.9	16757.0	21.3	16.3	0.2231	-0.6515
1695	23637.8	16727.8	22.0	17.0	0.2756	-0.5597
1696	23654.9	16735.8	22.3	17.3	0.1214	-0.9158
1697	23671.4	16743.6	20.7	15.7	0.1575	-0.8028
1868	22183.8	17064.4	33.6	28.6	0.4265	-0.3701
T-1	23552.8	16398.8			0.2200	-0.6576
T-5	23695.7	16343.8			0.4394	-0.3572
T-7	23550.0	16299.7			0.1043	-0.9817
T-10	23401.1	16251.0			0.4110	-0.3861
T-17	23550.5	16199.9			1.2983	0.1134
T-18	23653.4	16201.3			0.4564	-0.3407
T-20	23750.5	16201.0			0.4082	-0.3891
T-22	23405.9	16137.5			0.1301	-0.8857
T-27	23253.9	16098.9			0.2801	-0.5527
T-36	23455.1	15998.8			0.4110	-0.3861
1036	24231.1	16328.6			10.5164	1.0219
1037	24175.7	16228.1			10.4030	1.0172
1038	24096.9	16251.4			7.9086	0.8981
1039	24093.3	16313.3			8.3054	0.9194
1040	24103.1	16373.1			7.3133	0.8641
ETF-17	23606.4	16782.9			0.0272	-1.5648
ETF-25	23637.9	16773.9			0.2160	-0.6656
ETF-36	23658.6	16749.1			0.0431	-1.3657
ETF-39	23654.9	16735.8			0.1210	-0.9171
ETF-40	23671.4	16743.6			0.1570	-0.8040

Data sources: Wells 371-1868 from R. H. Ketelle, ORNL; wells T-1 through TF-40 from Bechtel 1991, Appendix A, Table 3A.1.

Fig. E.5. Hydraulic conductivity estimates as a function of depth at the SWSA 6 site.

Spatial Correlation and Anisotropy of the Hydraulic Conductivity Field

In an effort to characterize the spatial distribution of hydraulic conductivity estimates, the shallow K values from Table E.2 were subjected to a geostatistical analysis. The Geo-EAS program (EPA 1988) was used to generate a variogram of the data, using 100-ft lags (plus one initial 50-ft lag), as shown in Fig. E.6. Superimposed on the data points are four models of the variogram, none of which produces a satisfactory fit, an expected result given the wide scatter in the plot. It is clear that no spatial correlation can be justified in the K field. Additional variograms were generated to assess oriented correlation, since the K field might be expected to correlate better along geologic strike than across it. These directional variograms, however, showed no more spatial correlation than the variogram shown in Fig. E.6. This information supports the interpretation of the weathered regolith over SWSA 6 as a fairly uniform material.

Since no spatial correlation in the K field for SWSA 6 is to be found, no basis for generating a continuous field interpolated from point data exists (Journel and Huijbregts 1978). For that reason, a single value for hydraulic conductivity is appropriate for modeling purposes. This value was determined by calibration of the groundwater flow field to average measured water table elevations,

Anisotropy of the Hydraulic Conductivity Tensor

The hydraulic conductivity K discussed above is actually a tensor, $\mathbf{K}$, which varies with the direction in which it is applied. In the case of the 2-D MOC model, $\mathbf{K}$ is a 2-D tensor, with the major axis oriented along geologic strike. This orientation corresponds with "easting" (the x -axis) in the modeling grid. The degree to which the major axis dominates the minor is referred to as anisotropy of the $\mathbf{K}$ tensor (as opposed to any anisotropy of the K field), and is expressed as the ratio $K_x:K_y$. Values for the anisotropy of the hydraulic conductivity tensor were determined from Conasauga Group aquifer pump tests on the ORR. Estimates range from 3:1 to over 30:1 with maximum conductivity parallel to strike. (Davis et al. 1984; Lozier, Spiers, and Pearson 1987; Lee et al. 1989). A value of 3:1 was used for the base case of the PA, with 1:1, 3:1, and 30:1 used in the uncertainty analysis.

Partition Coefficients (K_d s)

Radionuclide specific data (half-life and decay coefficient) and chemical-specific data (soil/water partition coefficients) relevant to groundwater transport modeling are presented in Table E.3.

using K as the calibration variable.

Fig. E.6. Variogram of shallow hydraulic conductivity estimates at the SWSA 6 site.

Table E.3. Element-specific soil/water partition coefficients used for environmental transport analyses

Element	Soil/water K_d^a (mL/g)	Element	Soil/water K_d^a (mL/g)
H	0	I	0
Be	1000	Cs	3000
C	0	Sm	1000
Al	3000	Eu	40
Cl	0	Os	1000
K	30	Bi	500
Ca	30	Pb	100
Co	800	Ra	3000
Ni	2000	Ac	40
Se	0	Th	3000
Rb	30	Pa	400
Sr	30	U	40
Zr	50	Np	40
Nb	100	Pu	40
Tc	0	Am	40
Pd	2000	Cm	40
Cd	200	Cf	40
Sn	100		

^a Sources: See Sect. 2.6.3.5.

Retardation R is a function of K_d and the soil bulk density ρ_b and total porosity n , defined as

$$R = 1 + \frac{\rho_b K_d}{n} \quad (\text{E.14})$$

Retardation factors have a major influence on all components of the transport analysis. For the groundwater component, larger values of K_d effectively slowed the advective transport so that diffusion became an important factor for very large values for time.

E.2.2.2.4 Surface Recharge

Vertical recharge rates were assumed to vary spatially over the model domain. The values used in the analysis were taken directly from results for the digital terrain/UTM analyses described in Sect. 4.3.1.1.2. These analyses identified three primary recharge zones, which along with their corresponding recharge rates, are replicated in Table E.4. Additional factors such as the type of

disposal unit, the presence of engineered barriers such as caps, and the type of surface coverage (i.e., grass, gravel, etc.) were assumed to affect the recharge rates beneath the individual disposal units within SWSA 6. Additional UTM analyses were conducted in order to determine these recharge rates for all disposal units considered in the analysis. Fig. 4.11 shows the location and magnitude of these "modified" recharge values, along with the base recharge rates from Table E.4.

Table E.4. Model recharge rates

Model recharge zone	Recharge rate	
	(mm/year)	(in/year)
Stream channels	180	7.1
Ephemeral wetlands	140	5.5
Remaining landscape	90	3.5
Below-ground disposal units	175	6.9
Above-ground disposal units	628	25

E.2.2.2.5 Model Boundary Conditions

A number of physical features are addressed in the USGS MOC model through the use of specialized boundary conditions. These features include water table divides (modeled using inactive or no-flow boundaries), a proposed slurry wall near the high-range wells (modeled as no-flow cells), and creeks, other major surface water bodies, and the French drain located southeast of zone 8 (treated as constant head nodes by assigning large leakance values). Fig. E.7 shows the locations at which specialized boundary conditions were applied.

E.2.2.2.6 Groundwater Flow Model Calibration

Fig. E.8 shows the estimated water table configuration at SWSA 6 based on average observed water table elevations at wells located on the map. The water table contours shown are constrained by the ground surface elevations at perennial streams and by the invert elevation for the French drain. In preparing this contour map, it was observed that in all cases use of these features as constraints on water table elevation resulted in lowering the water table elevation in those areas. This characteristic indicates that in the average condition groundwater discharges to the streams and the French drain. Well water elevation responses to seasonal variations in precipitation and evapotranspiration vary depending on well location and depth. Observed water level fluctuations in wells at SWSA 6 range from less than 1.5 m (5 ft) to more than 3 m (10 ft).

Boundary Conditions for the SWSA 6 Groundwater Model

Fig. E.7. Boundary conditions used in the groundwater modeling of SWSA 6.

SWSA 6 Average Measured Head Elevations (ft)

Fig. E.8. Water table at SWSA 6, interpolated from average well measurements.

Given the material properties (Sect. E.2.2.2.3) for the porous medium and the recharge values imposed on the model (Sect. E.2.2.2.4), a field of water table head elevations can be calculated and compared with the average measured water table by interpolating between grid cell centers. By adjusting the hydraulic conductivity, this calculated water table can be brought into acceptable agreement with the measurements, and the model is said to be calibrated. The average measured water table elevations and those predicted by the calibrated groundwater flow model are presented in Table E.5. The calibrated water table elevations are displayed as a map in Fig. E.9.

The required velocities are obtained by solving the head equation subject to the constraint that the root-mean-square of the residuals between the computed and data-derived heads are minimal. This is an iterative process wherein the parameters and forcing terms are adjusted within reasonable limits. The MOC program is used to solve the differential equation for head on the 50×55 grid of block-centered cells. Given the yearly recharge rates supplied by the surface and shallow subsurface analysis in Sect. 4.3.1.1, a numerical solution for head was found by the program, and Darcy's law provided the bulk velocities from which the seepage velocity vectors were deduced, assuming fixed porosity. The steady-state groundwater velocity field for the area, shown in Fig. E.10, implies that groundwater starting at any point in the simulation area evidently reaches a stream, drain, or WOL. (Thus, the simulation does not provide for excursion of contaminated groundwater under WOL or otherwise off-site.)

E.2.2.3 Groundwater Contaminant Transport

Once the steady-state calibration of the flow model was completed, the next phase of the analysis was to use MOC to evaluate the transport in groundwater for each of the 63 radionuclides listed in Table 4.2. For all transport calculations, the effects of radioactive decay and retardation of contaminant due to linear adsorption were considered. This evaluation was performed by simultaneously (1) imposing the vertical component of the time-dependent release of contaminant from each disposal unit within SWSA 6 (calculated from the shallow subsurface transport analysis) onto the calibrated, steady-state flow field; and (2) performing a transient analysis for each nuclide to determine both the concentration in groundwater at proposed compliance points and the total contaminant flux discharged from groundwater to SWSA 6 surface waters. This flux was then combined with the lateral component of contaminant fluxes calculated from the shallow subsurface transport analysis to determine the total radionuclide concentrations in WOD at WOD. For each radionuclide, these transient calculations were repeated for several hypothetical inventories at the Interim Waste Management Facility (IWMF) in order to determine the disposal limits at IWMF for the drinking water pathway (see Sect. 4.3).

**Table E.5 Water table elevations used for calibration of the
SWSA 6 groundwater flow model.**

Well ID	Easting (ft)	Northing (ft)	Average measured water table elevation (ft)	Calibrated water table elevation (ft)	Difference in elevations (ft)
266	23327.0	15968.0	763.0	762.3	0.6
267	23221.0	16118.0	779.5	769.1	10.4
268	23307.0	16365.0	792.4	773.8	18.6
272	23520.0	16221.0	765.0	768.8	-3.8
273	23633.0	16126.0	758.1	761.7	-3.6
274	23347.0	16210.0	772.7	770.8	1.8
275	23737.0	16206.0	757.5	760.4	-2.9
276	23895.0	16211.0	756.6	756.6	0.0
277	24182.0	16207.0	757.4	758.4	-1.0
278	24223.0	16552.0	768.2	768.5	-0.2
279	24205.0	16696.0	773.5	772.4	1.1
284	23983.0	16750.0	773.8	773.3	0.6
292	23821.0	16773.0	775.9	776.2	-0.3
295	23992.0	16703.0	773.3	772.5	0.9
305	23914.0	16660.0	777.4	772.0	5.3
310	23939.0	16697.0	774.8	772.9	2.0
313	23605.0	16798.0	784.0	778.7	5.2
317	24328.0	16766.0	783.3	774.1	9.2
318	24323.0	17225.0	789.0	787.4	1.7
341	25165.0	16837.0	766.2	757.7	8.5
342	25154.0	16562.0	757.8	751.3	6.5
344	24810.0	16495.0	762.1	757.8	4.3
345	24881.0	16361.0	751.6	750.1	1.5
346	24631.0	16435.0	763.3	760.5	2.8
347	24604.0	16531.0	769.9	764.7	5.2
350	24649.0	16152.0	778.6	753.4	25.2
351	24661.0	16028.0	763.5	751.0	12.5
355	24991.0	16909.0	783.1	768.4	14.7
356	24451.0	16481.0	767.9	765.3	2.6
357	24599.0	16747.0	778.7	771.6	7.1
359	24867.0	16904.0	777.5	773.5	4.1
361	24166.0	16446.0	765.0	764.3	0.6
362	23718.0	16588.0	770.0	771.0	-1.0
363	23748.0	16420.0	762.8	762.9	-0.1
364	23838.0	16407.0	764.6	763.0	1.6
365	23681.0	16454.0	765.5	766.7	-1.1
367	25091.0	17723.0	790.9	784.9	6.0
368	25155.0	17348.0	788.8	772.6	16.2
368	25155.0	17348.0	794.7	772.6	22.1
369	24805.0	17476.0	808.0	792.1	15.9
369	24805.0	17476.0	802.9	792.1	10.8
370	24981.0	17145.0	786.0	777.0	9.0
371	25090.0	16393.0	746.6	748.3	-1.7

Table E.5. (continued)

Well ID	Easting (ft)	Northing (ft)	Average measured water table elevation (ft)	Calibrated water table elevation (ft)	Difference in elevations (ft)
372	25042.0	16570.0	759.6	754.4	5.2
373	25208.0	16945.0	762.0	758.4	3.5
374	25346.0	17462.0	763.5	761.8	1.7
376	23479.0	16580.0	785.5	775.2	10.3
377	23297.0	16610.0	795.1	777.2	17.9
378	23092.0	16474.0	789.6	776.5	13.1
381	24266.0	16248.0	758.4	759.6	-1.2
382	24025.0	15814.0	748.6	749.0	-0.4
383	24895.0	16165.0	746.4	746.4	0.0
384	24377.0	16131.0	754.8	756.7	-1.9
385	24669.0	17903.0	817.7	808.8	8.8
386	24828.0	16892.0	776.9	774.0	2.9
387	24886.0	16986.0	781.5	777.9	3.6
388	24886.0	17111.0	786.3	785.9	0.4
389	24203.5	16798.4	777.4	773.5	3.9
390	24199.7	16841.6	782.2	774.3	8.0
391	24192.4	16899.9	783.5	776.0	7.5
393	24259.0	16953.1	786.0	777.7	8.2
394	24420.1	17010.6	786.1	780.1	6.0
395	24235.3	16896.2	786.0	776.0	10.0
397	24227.3	16925.3	786.8	776.9	9.9
399	24349.5	16881.3	785.7	775.2	10.5
400	24304.6	17065.1	788.7	782.1	6.6
401	24380.2	17022.3	786.8	780.8	6.0
402	24277.5	16816.7	785.7	774.4	11.3
403	24163.1	16777.7	770.3	772.8	-2.5
404	24159.1	16786.3	769.7	772.8	-3.1
405	24158.0	16791.8	769.9	772.8	-3.0
636	24326.2	17668.0	804.8	802.9	1.9
639	24138.1	17395.4	796.3	790.8	5.5
640	24719.6	17614.7	802.2	797.0	5.2
644	24848.4	16748.8	769.8	765.0	4.8
645	25274.6	17173.7	757.7	760.3	-2.6
648	24524.2	17374.6	799.8	792.9	7.0
649	25075.5	17375.5	792.0	778.7	13.2
739	23604.0	15628.9	745.7	746.7	-1.0
831	24133.5	17387.4	795.3	790.6	4.7
835	23951.8	15767.7	746.0	747.5	-1.5
836	24138.8	15748.2	745.6	747.8	-2.2
837	24352.6	15845.6	747.7	750.1	-2.4
838	24934.8	16308.7	748.5	747.6	0.9
840	25251.7	16928.6	748.0	755.5	-7.5
842	25298.4	17216.1	759.0	760.1	-1.1
843	25221.4	17597.1	779.9	774.0	6.0
845	24988.3	17108.2	786.8	774.8	12.1

Table E.5. (continued)

Well ID	Easting (ft)	Northing (ft)	Average measured water table elevation (ft)	Calibrated water table elevation (ft)	Difference in elevations (ft)
848	24656.3	16942.4	779.0	778.2	0.8
850	24793.5	16595.4	765.3	762.0	3.2
852	23911.6	16346.9	761.4	761.0	0.3
853	24047.5	16695.1	771.3	771.7	-0.4
854	23851.9	16711.9	773.7	774.1	-0.4
857	23106.3	16538.0	792.0	777.2	14.8
859	23246.2	16009.4	762.5	764.8	-2.3
1036	24231.1	16328.6	762.5	761.5	1.0
1037	24175.7	16228.1	755.8	758.9	-3.1
1038	24096.9	16251.4	757.6	758.8	-1.3
1039	24093.3	16313.3	759.3	759.3	0.0
1040	24103.1	16376.1	759.4	760.6	-1.2
1157	25192.4	17807.0	797.9	782.6	15.2
1158	25102.5	17816.1	800.0	786.6	13.4
1159	25062.1	17752.3	800.9	787.1	13.8
1160	25002.5	17654.2	800.8	787.5	13.3
1161	24928.2	17549.8	799.6	788.5	11.1
1162	25086.4	17609.0	794.4	782.2	12.3
1163	25141.2	17689.5	795.2	781.4	13.8
1165	25041.8	17879.5	787.7	788.8	-1.1
1166	24945.9	17697.2	795.8	791.2	4.6
1167	25256.0	17709.8	783.1	775.6	7.5
1168	25295.2	17645.7	777.3	770.8	6.5
1169	25151.4	17895.1	783.3	785.4	-2.1
1225	24603.7	17754.8	817.0	808.8	8.2
1227	23838.2	16569.4	768.3	768.9	-0.7
1228	23878.8	16334.7	760.3	760.8	-0.5
1229	24072.7	16453.3	761.4	762.1	-0.7
1232	24218.9	17020.1	788.3	779.8	8.4
1233	24211.9	17005.3	782.4	779.3	3.1
1242	25344.8	17367.9	760.1	760.4	-0.4
1243	25239.0	17085.6	760.1	759.8	0.3
1257	24251.2	16493.3	766.7	766.5	0.2
1659	23641.2	16756.0	783.0	777.5	5.4
1660	23665.2	16778.3	782.3	777.9	4.4
1661	23672.8	16764.9	780.1	777.5	2.6
1662	23673.3	16749.1	781.4	777.1	4.3
1663	23665.3	16732.5	781.2	776.7	4.5
1664	23651.0	16724.1	781.1	776.6	4.4
1665	23633.6	16723.2	781.4	776.8	4.6
1668	23609.8	16763.5	783.1	778.0	5.1
1675	23606.4	16782.0	787.6	778.4	9.2
1676	23624.8	16791.2	787.6	778.5	9.2
1680	23641.2	16788.0	783.6	778.3	5.3
1683	23637.9	16773.9	782.9	778.0	4.9

Table E.5. (continued)

Well ID	Easting (ft)	Northing (ft)	Average measured water table elevation (ft)	Calibrated water table elevation (ft)	Difference in elevations (ft)
1685	23616.4	16745.5	782.8	777.5	5.3
1687	23651.4	16769.4	781.9	777.8	4.1
1693	23658.6	16749.1	782.1	777.2	4.9
1696	23654.9	16735.8	781.8	776.9	4.9
1697	23671.4	16743.6	781.3	777.0	4.4
1717	24375.0	16002.0	747.6	753.8	-6.2
1718	24541.0	15872.0	748.1	749.2	-1.1
1719	24331.0	15762.0	746.8	747.8	-1.0
1720	24409.0	15882.0	749.7	750.7	-1.0
1721	24405.0	15867.0	749.8	750.3	-0.5
1722	24393.0	15855.0	749.8	750.1	-0.3
1723	24378.0	15851.0	749.6	750.1	-0.5
1724	24363.0	15856.0	750.0	750.3	-0.3
1725	24352.0	15867.0	751.6	750.6	0.9
1726	24348.0	15881.0	750.4	751.0	-0.6
1727	24379.0	15882.0	751.5	750.9	0.6
1728	24377.0	15946.0	749.2	752.5	-3.3
1863	23847.0	16687.0	772.5	773.3	-0.7
1864	23734.0	16679.0	776.0	773.9	2.1
1865	23782.0	16689.0	773.1	773.6	-0.5
2385	24474.9	16527.3	770.7	766.8	3.9
2388	24178.2	16810.9	772.2	773.4	-1.2
2390	24467.9	16674.2	772.2	772.1	0.1
2391	24467.0	16696.0	772.7	773.3	-0.5
2392	24464.5	16724.1	774.9	774.9	0.1
2393	24458.0	16773.1	775.0	775.0	0.0
2394	24452.4	16818.3	775.0	774.3	0.8
2395	24449.0	16840.9	774.9	773.9	1.0
2396	24446.0	16870.6	775.0	773.5	1.5
2397	24443.2	16893.8	775.0	773.2	1.8
2398	24440.8	16922.8	775.0	773.5	1.5
2399	24438.0	16943.3	775.0	774.4	0.5
2451	24424.3	17017.6	785.0	780.7	4.3
2452	24432.5	16927.8	777.9	774.1	3.8
2469	24620.8	17627.7	802.5	799.4	3.1
2599	24469.2	16625.3	771.6	772.0	-0.4
2607	25150.6	17273.9	779.6	769.4	10.2

Data sources: The X-10 Wells Database and EBASCO 1992.

SWSA 6 Calibrated Head Elevations (ft)

Fig. E.9. Modeled water table at SWSA 6, calibrated to well measurements.

SWSA 6 Calibrated Seepage Velocities

Fig. E.10. Modeled groundwater seepage velocities at SWSA 6.

E.2.2.3.1 Time-Dependent Contaminant Loading

As indicated previously, groundwater contaminant transport calculations were performed using the vertical component of the time-dependent contaminant fluxes calculated during the shallow subsurface transport analysis by the PADSIM/HOLSIM codes as input. The contaminant flux rates for each of the disposal units within SWSA 6 were converted to a "recharge concentration" based on the volumetric recharge rates associated with the unit (see Sect. E.2.2.2.4) and introduced simultaneously into the groundwater flow model.

The fluxes calculated by PADSIM for the IWWMF represent the release of contaminant from any one of the unit's disposal pads. These six pads were assumed to be filled at the rate of one pad every 18 months. As a result, it would be possible for some IWWMF pads to release contamination before others are loaded. For short-lived, highly mobile nuclides, this irregular loading pattern could have a considerable effect on the time, location, and magnitude of ground and surface water concentrations. Therefore, in order to reflect a loading rate at IWWMF of one pad every 18 months, the fluxes calculated by PADSIM were applied at each pad location on a staggered basis.

E.2.2.3.2 Location of Compliance Points

One of the primary goals of the contaminant transport analysis was to determine disposal limits at IWWMF such that the maximum groundwater concentrations for each nuclide at designated receptor locations were in accordance with the 4 mrem per year dose limit presented in Sect. 2. For purposes of this analysis, two sets of receptor locations were considered. The first corresponded to grid cells located 100 m from all disposal units within SWSA 6. Fig. 4.15 shows the location of these model cells. The second set of receptor locations, shown in Fig. 4.16, corresponds to the expanded buffer zone described in Sect. 4.3.1.

E.3 MODIFICATIONS TO THE USGS MOC PROGRAM CODE

The modeling of contaminant transport in groundwater for the SWSA 6 site was a complicated process involving long-term simulations with expansive multiple plumes, as well as incorporation of site-specific details, such as spatially variable recharge rates. Furthermore, the Latin hypercube (LHC) analysis (Sect. 5.4) required that multiple simulations be conducted for each radionuclide analyzed. Since the ability to perform these functions was beyond that of the USGS MOC Version 3.0 code, in-house modifications were necessary.

E.3.1 Description of Code Modifications

The chief modifications made at ORNL to the USGS MOC 3.0 computer code can be summarized as follows:

1. Most of the array dimensions found in the original code were increased. This increase was made primarily to allow larger model grids (the grid size of the original MOC code was only 40×40) and more observation nodes (used for computing concentrations and fluxes at compliance points).
2. All floating-point and integer model inputs were converted to free format.
3. Numerous output statements were added to the code in which relevant various model results were reformatted or summarized. For example, files listing yearly values for total contaminant discharged to surface water locations were generated.
4. The LHC analyses described in Sect. 5.4 and Appendix I required modifications to enable runs using multiple parameter sets as input. Additional modifications were made for storing and summarizing the LHC results.
5. In addition to the original MOC input file, incorporation of item 4 above required the creation of two additional input files:
 - (a) multiple input files containing time-dependent contaminant concentrations in recharge for each disposal unit at SWSA 6
 - (b) an LHC parameter file, containing the multiple sets of input parameters and associated control variables required for performing LHC analyses.

The LHC file consisted of parameters describing the number of LHC iterations to be performed, the number of simulation years for each iteration, the number of and associated iterations for which output is generated, scale factors used to modify values for the LHC input parameters (hydraulic conductivity, distribution coefficient, etc.), the number and names of input files containing time-dependent concentrations of contaminant in recharge, and finally, the values for the input parameters considered in the LHC analysis for each LHC iteration.

All of the above modifications were written in FORTRAN. The ORNL version of the USGS MOC code was compiled and run on an HP 9000, series 730 workstation.

E.3.2 Code Verification

To verify that the modifications made to enable MOC to perform LHC analyses were performing as expected, deterministic model runs were made using representative sets of input parameters from the LHC analysis. In all cases, results from the deterministic analysis matched those from the corresponding LHC sample.

E.4 REFERENCES

Bear, J. 1972. *Dynamics of Fluids in Porous Media*, Dover Publications, New York

- Bechtel (Bechtel National, Inc.) 1991. *RCRA Facility Investigation Report for Waste Area Grouping 6 at Oak Ridge National Laboratory, Oak Ridge, Tennessee*, Vol. 3, ES/ER-22/V3 & D1, ORNL/ER/Sub-87/99053/5/V3, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Davis, E. C., et al. 1984. *Site Characterization Techniques Used at a Low-Level Waste Shallow Land Burial Field Demonstration Facility*, ORNL/TM-9146, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- EBASCO (EBASCO Services Inc.) 1992. *OR-ERWM Program Design Criteria Studies: Results of 3DFEMWATER Modeling*, ER W6.01-002, Rev. 0, Environmental Consulting Engineers, Inc., Knoxville, Tenn., November 23.
- EPA (U.S. Environmental Protection Agency) 1988. Geo-EAS (Geostatistical Environmental Assessment Software), computer program by E. Englund and A. Sparks, EPA/600/4-88/033a, Washington, D.C.
- Journel, A. G., and Ch. J. Huijbregts 1978. *Mining Geostatistics*, Academic Press, New York.
- Konikow, L. F., and J. D. Bredehoeft 1978. USGS MOC 1990 Computer Code, with instructions in "Computer Model of Two-Dimensional Solute Transport and Dispersion in Ground Water," chapter C2, book 7, of *Geological Survey Techniques of Water-Resources Investigations of the United States Geological Survey*, U.S. Department of the Interior, Washington, D.C.
- Lee, R. R., et al. 1989. *Calibration of a Groundwater Flow and Containment Transport Model: Progress Toward Model Validation*, ORNL/TM-11294, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Lozier, W. B., C. A. Spiers, and R. Pearson 1986. *Aquifer Pump Test with Tracers*, ORNL/Sub/86-32136/2, Martin Marietta Energy Systems, Inc., Oak Ridge Natl. Lab.

Appendix F

PREDICTED AMOUNTS OF RADIONUCLIDES LEACHED FROM SWSA 6 DISPOSAL UNITS AND CORRESPONDING SURFACE WATER CONCENTRATIONS AT WHITE OAK DAM

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Appendix F

PREDICTED AMOUNTS OF RADIONUCLIDES LEACHED FROM SWSA 6 DISPOSAL UNITS

This section presents additional results from the environmental pathways transport analysis. These are designed to provide the reader with additional insight regarding the individual contribution from the various disposal units within SWSA 6 to the total radionuclide concentration in surface water at White Oak Dam (WOD). All the results presented in this appendix assume that the inventories of inactive facilities are those presented in Table 4.6, and that the inventory in the IWMF is that allowable inventory based on the expanded buffer zone scenario, corresponding to the inventories presented in Table 5.27. Note that the peak concentrations in that table are in the groundwater, while those presented in this appendix are in the surface water at WOD.

F.1 DISPOSAL FACILITY RELEASE ESTIMATES

Releases from the various disposal units in SWSA 6 are calculated by the SOURCE1 and SOURCE2 programs, the results of which are discussed in Sect. 5.1.1.1. Tables in that section provide peak mass fluxes for all radionuclides from each disposal facility analyzed in the PA. Much insight can be gained, however, by examining the behavior of the releases through time, rather than just the peak values. Given the number of radionuclides analyzed (63) and the number of waste facilities included (12 inactive facilities plus the IWMF), it is impractical to provide the reader with a comprehensive set of release curves.

The behavior of the radionuclides is captured by a subset of the 63, containing the following ten radionuclides: ^3H , ^{14}C , ^{90}Sr , ^{99}Tc , ^{137}Cs , ^{232}Th , ^{233}U , ^{238}U , ^{239}Pu , and ^{241}Am . These were chosen to represent a wide range of the major environmental transport characteristics: half-life, K_d , and solubility. Mass release rates for these ten nuclides from all modeled facilities are provided in the figures in this appendix, interleaved with corresponding graphs of surface water concentrations.

F.2 SURFACE WATER CONCENTRATION ESTIMATES

The following figures are graphs of surface water concentration at WOD vs time, with contributions from each modeled disposal facility within SWSA 6, for the selected subset of ten

radionuclides. These figures are useful for identifying the primary contributors to dose. Abbreviations used for SWSA 6 disposal facilities in each of these figures are presented in Table F.1.

Table F.1 Abbreviations used for SWSA 6 disposal facilities

Disposal Facility	Abbreviation
High-Range Silos	Zone 03
High-Range Wells	Zone05 wh
High-Range Wells-in-Silos	Zone05 whs
Fissile Wells	Zone 07
Quadrex Trenches	Zone 08 tq
Low-Range Silos	Zone 08 tl
Biological Trenches	Zone 15
Asbestos Silos	Zone 16
Biological Trenches	Zone 20
IWMF	Zone 25
Tumulus II	Zone 26
Tumulus I	Zone 27
Low-Range Silos	Zone 30

Fig. F.1. Calculated release rates of ^3H from SWSA 6 disposal sites.

Fig. F.2. Calculated zone-specific contributions to surface water concentrations at WOD for ^3H .

Fig. F.3. Calculated release rates of ^{14}C from SWSA 6 disposal sites.

Fig. F.4. Calculated zone-specific contributions to surface water concentrations at WOD for ^{14}C .

Fig. F.5. Calculated release rates of ^{90}Sr from SWSA 6 disposal sites.

Fig. F.6. Calculated zone-specific contributions to surface water concentrations at WOD for ^{90}Sr .

Fig. F.7.7. Calculated release rates of ⁹⁹Tc from SWSA 6 disposal sites.

Fig. F.8. Calculated zone-specific contributions to surface water concentrations at WOD for ^{99}Tc .

Fig. F.9. Calculated release rates of ^{137}Cs from SWSA 6 disposal sites.

Fig. F.10. Calculated zone-specific contributions to surface water concentrations at WOD for ^{137}Cs .

Fig. F.11. Calculated release rates of ^{232}Th from SWSA 6 disposal sites.

Fig. F.12. Calculated zone-specific contributions to surface water concentrations at WOD for ^{232}Th .

Fig. F.13. Calculated release rates of ^{233}U from SWSA 6 disposal sites.

Fig. F.14. Calculated zone-specific contributions to surface water concentrations at WOD for ^{233}U .

Fig. F.15. Calculated release rates of ^{238}U from SWSA 6 disposal sites.

Fig. F.16. Calculated zone-specific contributions to surface water concentrations at WOD for ^{238}U .

Fig. F.17. Calculated release rates of ^{239}Pu from SWSA 6 disposal sites.

Fig. F.18. Calculated zone-specific contributions to surface water concentrations at WOD for ^{239}Pu .

Fig. F.19. Calculated release rates of ²⁴¹Am from SWSA 6 disposal sites.

Fig. F.20. Calculated zone-specific contributions to surface water concentrations at WOD for ^{241}Am .

Appendix G

DOSE ANALYSIS FOR OFF-SITE INDIVIDUALS AND INADVERTENT INTRUDERS

D. C. Kocher

Appendix G

DOSE ANALYSIS FOR OFF-SITE INDIVIDUALS AND INADVERTENT INTRUDERS

G.1 INTRODUCTION

This appendix presents the models and data bases used to estimate radiation doses to individuals per unit concentration of radionuclides in various exposure media following disposal of low-level waste in the different types of disposal units in the SWSA 6 facility. The dose analysis considers two groups of exposed individuals:

- *off-site individuals*, i.e., members of the public who are assumed to be located outside the boundary of the disposal site; and
- *inadvertent intruders*, i.e., individuals who are assumed to come onto the disposal site following the loss of active institutional controls.

Off-site individuals may receive exposures from radionuclides transported beyond the site boundary at any time after disposal. However, in accordance with current Department of Energy (DOE) policy (DOE 1988a, 1996), inadvertent intrusion onto the disposal site is assumed to be precluded by active institutional controls for at least 100 years after disposal.

The performance assessment for low-level waste (LLW) disposal in the SWSA 6 facility assumes that groundwater and surface water pathways are the primary means for the transport of radionuclides from disposal units into the environment beyond the boundary of the disposal site. Therefore, off-site individuals are assumed to receive radiation doses as a result of exposure to contaminated groundwater or surface waters.

The dose analysis for inadvertent intruders considers four different exposure scenarios involving direct intrusion into solid waste in disposal units. In accordance with current DOE policy (DOE 1996), scenarios involving exposure of inadvertent intruders to contaminated groundwater or surface water on the disposal site are not considered, because allowable releases of radionuclides in water are determined by performance objectives for off-site releases.

The following sections discuss the assumed exposure scenarios and exposure pathways for off-site individuals or inadvertent intruders and the models and parameter values used in calculating doses for each exposure scenario and pathway. For exposures of off-site individuals to contaminated water, the results of the dose analysis are presented in the form of annual effective dose equivalents per unit concentration of radionuclides in water. For exposures of inadvertent intruders resulting from direct intrusion into disposal units, the results are presented in the form of annual effective dose

equivalents or total effective dose equivalents per unit concentration of radionuclides in disposal units, depending on whether the scenario involves chronic or short-term exposure.

G.2 RADIONUCLIDES CONSIDERED IN DOSE ANALYSIS

As indicated in Appendix A, LLW previously sent to SWSA 6 for disposal has contained a large number of radionuclides. However, not all of these radionuclides are potentially important in a dose analysis for off-site individuals or inadvertent intruders. Many of these radionuclides have occurred only in very low concentrations, and others are sufficiently short-lived that they would decay to innocuous levels before off-site transport in water or inadvertent intrusion onto the disposal site could occur.

The radionuclides considered in the dose analyses for off-site individuals and inadvertent intruders are listed in Table G.1. The choice of radionuclides was based on the following considerations. First, all radionuclides with half-lives of about 5 years or greater that were reported in significant quantities in previous disposals in SWSA 6 (see Appendix A) are included. As noted above, radionuclides with half-lives less than about 5 years could not result in significant doses to off-site individuals or inadvertent intruders and, thus, are not included in the dose analyses. Second, other radionuclides with half-lives greater than 5 years are included if they could arise from operations of the nuclear fuel cycle or the production of nuclear weapons, including production in nuclear fission or by neutron activation of materials that would be present in reactors, because these radionuclides are likely to be the most important sources of radioactive wastes on the Oak Ridge Reservation (ORR). If additional radionuclides from other sources or processes are found to be present in wastes intended for disposal in SWSA 6, they can easily be included in the dose analysis.

G.3 ASSUMED EXPOSURE SCENARIOS AND EXPOSURE PATHWAYS

This section describes the exposure scenarios and exposure pathways assumed in the dose analyses for off-site individuals and inadvertent intruders. The model equations and parameter values for each exposure pathway are presented in Sect. G.5, except the dosimetry data used in the analyses are presented in Sect. G.4.

G.3.1 Exposure Scenarios and Exposure Pathways for Off-Site Individuals

The performance assessment for disposal units in the SWSA 6 facility assumes that radionuclides are transported to locations beyond the facility boundary primarily via the groundwater and surface water pathways. Based on historical data on airborne releases from all sources on the

Table G.1. Radionuclides considered in dose analyses for off-site individuals and inadvertent intruders

Nuclide ^a	Half-life ^b	Nuclide ^a	Half-life ^b
³ H	12.28 y	¹⁵¹ Sm	90 y
¹⁰ Be	1.6×10^6 y	¹⁵² Eu	13.6 y
¹⁴ C	5730 y	¹⁵⁴ Eu	8.8 y
²⁶ Al	7.2×10^5 y	¹⁵⁵ Eu	4.96 y
³⁶ Cl	3.01×10^5 y	¹⁹⁴ Os	6.0 y ^d
⁴⁰ K	1.277×10^9 y	²⁰⁷ Bi	33.4 y
⁴¹ Ca	1.03×10^5 y	²¹⁰ Pb	22.26 y
⁶⁰ Co	5.271 y	²¹⁰ Po (1.0)	138.378 d
⁵⁹ Ni	7.5×10^4 y	²²⁶ Ra	1600 y
⁶³ Ni	100.1 y	²²² Rn (1.0)	3.8235 d
⁷⁹ Se	6.5×10^4 y	²¹⁴ Pb (1.0)	26.8 m
⁸⁷ Rb	4.73×10^{10} y	²¹⁴ Bi (1.0)	19.9 m
⁹⁰ Sr	28.6 y	²¹⁰ Pb (1.0) ^c	
⁹⁰ Y (1.0)	64.1 h	²²⁸ Ra	5.75 y
⁹³ Zr	1.53×10^6 y	²²⁸ Ac (1.0)	6.13 h
^{93m} Nb (1.0) ^c		²²⁸ Th (1.0)	1.9132 y
^{93m} Nb	14.6 y	²²⁴ Ra (1.0)	3.62 d
⁹⁴ Nb	2.03×10^4 y	²²⁰ Rn (1.0)	55.61 s
⁹⁹ Tc	2.13×10^5 y	²¹² Pb (1.0)	10.643 h
¹⁰⁷ Pd	6.5×10^6 y	²¹² Bi (1.0)	60.55 m
^{113m} Cd	13.7 y	²⁰⁸ Tl (0.3593)	3.053 m
^{121m} Sn	55 y	²²⁷ Ac	21.773 y
¹²⁶ Sn	1.0×10^5 y	²²⁷ Th (0.9862)	18.718 d
^{126m} Sb (1.0)	19.0 m	²²³ Ra (1.0)	11.434 d
¹²⁶ Sb (0.14)	12.4 d	²¹¹ Pb (1.0)	36.1 m
¹²⁹ I	1.57×10^7 y	²¹¹ Bi (1.0)	2.13 m
¹³⁵ Cs	2.3×10^6 y	²⁰⁷ Tl (0.99727)	4.77 m
¹³⁷ Cs	30.17 y		
^{137m} Ba (0.946)	2.552 m		

Table G.1. (continued)

Nuclide ^a	Half-life ^b	Nuclide ^a	Half-life ^b
²²⁹ Th	7.34×10^3 y	²³⁹ Pu	24131 y
²²⁵ Ra (1.0)	14.8 d	²⁴⁰ Pu	6569 y
²²⁵ Ac (1.0)	10.0 d	²⁴¹ Pu	14.4 y
²²¹ Fr (1.0)	4.8 m	²⁴² Pu	3.758×10^5 y
²¹³ Bi (1.0)	45.65 m	²⁴⁴ Pu	8.26×10^7 y
²⁰⁹ Tl (0.0216)	2.20 m	^{240m} Np (1.0)	7.4 m
²³⁰ Th	7.7×10^4 y	²⁴¹ Am	432.2 y
²³² Th	1.405×10^{10} y	^{242m} Am	152 y
²²⁸ Ra (1.0) ^c		²⁴² Am (0.99524)	16.02 h
²³¹ Pa	3.276×10^4 y	²⁴² Cm (0.823)	163.2 d
²²⁷ Ac (1.0) ^c		²³⁸ Np (0.00476)	2.117 d
²³² U ^e	72 y	²³⁸ Pu (0.827) ^c	
²³³ U	1.5952×10^5 y	²⁴³ Am	7380 y
²³⁴ U	2.445×10^5 y	²³⁹ Np (1.0)	2.355 d
²³⁵ U	7.038×10^8 y	²⁴³ Cm	28.5 y
²³¹ Th (1.0)	25.52 h	²⁴⁴ Cm	18.11 y
²³⁶ U	2.3415×10^7 y	²⁴⁵ Cm	8.5×10^3 y
²³⁸ U	4.468×10^9 y	²⁴⁶ Cm	4.75×10^3 y
²³⁴ Th (1.0)	24.10 d	²⁴⁷ Cm	1.56×10^7 y
^{234m} Pa (1.0)	1.17 m	²⁴³ Pu (1.0)	4.956 h
²³⁴ Pa (0.0016)	6.70 h	²⁴⁸ Cm	3.39×10^5 y
²³⁷ Np	2.14×10^6 y	²⁴⁹ Cf	350.6 y
²³³ Pa (1.0)	27.0 d	²⁵⁰ Cf	13.08 y
²³⁸ Pu	87.85 y	²⁵¹ Cf	900 y

^aIndented entries are radiologically significant shorter-lived decay products of parent radionuclide listed. For each decay product, branching fraction in decay of parent radionuclide obtained from Kocher (1981) is given in parentheses.

^bValues from Kocher (1981), except as noted. Units are y = years, d = days, h = hours, m = minutes, s = seconds.

^cRadionuclide and its decay products are listed separately when half-life is greater than 5 years and radionuclide could exist in disposed waste separate from longer-lived parent.

^dHalf-life from Tuli (1995).

^eShorter-lived decay products ²²⁸Th, ²²⁴Ra, ²²⁰Rn, ²¹²Pb, ²¹²Bi, and ²⁰⁸Tl are listed following entry for ²²⁸Ra.

ORR off-site transport via the atmospheric pathway following suspension into the air of radionuclides in solid waste is assumed to be relatively unimportant. Potential exposures of off-site individuals from atmospheric releases are considered further in Appendix H.

Off-site individuals are assumed to use contaminated groundwater for domestic purposes, and the following exposure pathways are assumed to occur:

- direct ingestion of contaminated drinking water, and
- ingestion of milk and meat obtained from dairy and beef cattle that drink contaminated water.

Use of contaminated groundwater by off-site individuals for irrigation of food crops or pasture land, which would result in additional exposure pathways, is not considered to be a credible occurrence, because rainfall typically is abundant in Oak Ridge and irrigation is not commonly practiced near the disposal site at the present time.

Off-site individuals are assumed to use contaminated surface water for domestic and recreational purposes, and the following exposure pathways are assumed to occur:

- direct ingestion of contaminated drinking water,
- ingestion of milk and meat obtained from dairy and beef cattle that drink contaminated water,
- ingestion of fish obtained from contaminated water, and
- external exposure while swimming in contaminated water.

The first two pathways are the same as those from the use of contaminated groundwater, and the two additional pathways are credible only for the use of surface water. For the reasons described above, the use of contaminated surface water by off-site individuals for the irrigation of food crops or pasture land is not considered to be a credible occurrence.

G.3.2 Exposure Scenarios and Exposure Pathways for Inadvertent Intruders

Inadvertent intruders are assumed to come onto the disposal site at any time after loss of active institutional controls over the site at 100 years after disposal, and without prior knowledge of waste disposal activities at the site. Inadvertent intruders then are assumed to receive radiation exposures from direct intrusion into solid waste in the various disposal units. In accordance with current DOE policy (DOE 1996), exposure scenarios involving the use of contaminated groundwater or surface water on the disposal site are not considered in the dose analysis for inadvertent intruders.

For direct intrusion into the various disposal units in the SWSA 6 facility after the loss of active institutional control over the disposal site, exposures are assumed to occur according to one of four different scenarios: the *agriculture*, *resident*, *discovery*, and *post-drilling* scenarios. In all scenarios, an intruder is assumed to reside on the disposal site and to establish on-site sources of foodstuffs. The four exposure scenarios and their associated exposure pathways are described as follows.

In the *agriculture* scenario, an inadvertent intruder is assumed to construct a home on top of disposal units, with the foundation extending directly into exposed waste. In addition, some of the radioactive wastes and other waste materials exhumed during construction of the foundation for the home, which are assumed to be indistinguishable from native soil, are assumed to be mixed with native soil in the intruder's vegetable garden. Based on these assumptions, the following pathways involving exposure to radionuclides in solid waste are assumed to occur:

- ingestion of vegetables grown in contaminated garden soil,
- direct ingestion of contaminated soil from the garden in conjunction with vegetable intakes,
- external exposure to contaminated soil while working in the garden or residing in the home on top of the disposal units, and
- inhalation of radionuclides suspended into the air from contaminated soil while working in the garden or residing in the home.

In the *resident* scenario, an inadvertent intruder is assumed to dig a foundation for a home at the location of disposal units, as in the agriculture scenario described above, but is assumed to encounter an intact and impenetrable engineered barrier (e.g., reinforced concrete) on top of the disposal units. The intruder then is assumed to construct a home directly on top of the intact engineered barrier above the waste. Because the engineered barrier is assumed not to be breached during excavation, the only exposure pathway of concern for the resident scenario is external exposure to photon-emitting radionuclides in the waste during the time the intruder resides in the home on top of the disposal units.

Because of the presence of an engineered barrier in the resident scenario, which provides considerable shielding, the external dose per unit concentration of radionuclides in disposed waste while residing in the home on top of disposal units will always be less in the resident scenario than in the agriculture scenario. Therefore, the resident scenario is intended to be applied only when inadvertent intrusion onto the disposal site first becomes a credible occurrence at 100 years after disposal and the engineered barriers presumably are intact. At this time, the external dose from shorter-lived radionuclides in the waste could be considerably higher than the dose at later times when the engineered barrier has degraded and residence on top of exposed waste could occur but the concentrations of the shorter-lived radionuclides would be greatly reduced by decay. The resident scenario is not relevant for disposal units constructed without engineered barriers.

In the *discovery* scenario, an inadvertent intruder is assumed (1) to dig a foundation for a home at the location of disposal units and (2) to encounter an intact and impenetrable engineered barrier (e.g., reinforced concrete) used in constructing the units, as in the resident scenario described above. However, shortly after encountering the engineered barrier, the intruder decides to abandon digging at that location and moves elsewhere. Since waste in the disposal units is not directly accessed during excavation, because of the assumed impenetrability of the engineered barrier, the only exposure pathway for this scenario is external exposure to photon-emitting radionuclides in the disposal units during the time the intruder digs at that location.

For a given thickness of an engineered barrier, the external dose per unit concentration of radionuclides will always be greater in the resident scenario than in the discovery scenario, because of the much longer exposure time for the resident scenario. However, some disposal units in the SWSA 6 facility are constructed with engineered barriers at the top that are considerably thicker than the engineered barriers at the sides. Therefore, the discovery scenario is intended to take into account that, in excavating at the disposal site, an inadvertent intruder could approach disposal units from the side, rather than the top, and the lesser exposure time in the discovery scenario could be compensated by the reduced shielding at the side of disposal units. The discovery scenario presumes that an intruder would not construct a home immediately beside engineered disposal units. The discovery scenario also is not relevant for disposal units constructed without engineered barriers.

In the *post-drilling* scenario, direct intrusion into disposal units by excavation is assumed not to occur during construction of a home. Direct intrusion into disposal units during construction could be precluded by the presence of impenetrable engineered barriers, by burial of waste at depths greater than the normal depth of a foundation for a home, or by the construction of a home at a location away from disposal units. However, an inadvertent intruder is assumed to access solid waste by drilling through a disposal unit (e.g., for the purpose of constructing a well for a domestic water supply) and, during drilling, a small volume of radioactive waste, which is assumed to be indistinguishable from native soil, is brought to the surface. All of the drilling waste then is assumed to be mixed with native soil in the intruder's vegetable garden, and the following pathways involving exposure to radionuclides in the solid waste are assumed to occur:

- ingestion of vegetables grown in contaminated garden soil,
- direct ingestion of contaminated soil from the garden in conjunction with vegetable intakes,
- external exposure to contaminated soil while working in the garden, and
- inhalation of radionuclides suspended into air from contaminated soil while working in the garden.

These pathways correspond to some of the pathways assumed for the agriculture scenario. However, in the post-drilling scenario, external and inhalation exposures while residing in the home are not considered, because all of the drilling waste is assumed to be mixed with soil in the vegetable garden.

G.4 DOSE CONVERSION FACTORS FOR INTERNAL AND EXTERNAL EXPOSURE

From the descriptions of the assumed exposure scenarios given above, doses to off-site individuals or inadvertent intruders are assumed to result from ingestion and inhalation of radionuclides and from external exposure to photons emitted by radionuclides in contaminated water, contaminated soil, or in the disposal units themselves. This section presents the factors used in the

dose analyses to convert (1) intakes of radionuclides via ingestion and inhalation to internal doses and (2) concentrations of radionuclides in water, soil, or disposal units to external dose rates.

The internal dose conversion factors for ingestion and inhalation of radionuclides used in this analysis are given in Tables G.2 and G.3, respectively. The internal dose conversion factors give 50-year committed effective dose equivalents per unit activity intake of a radionuclide. These values were developed by the International Commission on Radiological Protection (ICRP), and have been adopted by Federal agencies for use in assessing radiation doses to the public (Eckerman et al. 1988). Dose conversion factors for inhalation of isotopes of radon and their short-lived decay products are not listed in Table G.3, but are calculated using a natural analog model described in Sect. G.5.2.1.

For some radionuclides, more than one dose conversion factor for ingestion or inhalation is given by Eckerman et al. (1988). If ingestion dose conversion factors are given for two values of the GI-tract absorption fraction (f_1), the dose conversion factor corresponding to the higher absorption fraction is adopted in Table G.2, because radionuclides that are transported in water or through terrestrial food chains should be in relatively soluble form and more easily absorbed in the GI tract. In all cases, this choice gives the higher dose conversion factor. If inhalation dose conversion factors are given for more than one lung clearance class (D, W, or Y), the clearance class giving the highest dose conversion factor is adopted in Table G.3 for most radionuclides. However, ^{90}Sr and ^{99}Tc are assumed to be Class D, because their solubility in the environment is expected to be relatively high, and all isotopes of thorium are assumed to be Class Y, because thorium in soil is expected to be highly insoluble (e.g., in oxide or hydroxide form). On the other hand, the particular choices of inhalation dose conversion factors for ^{90}Sr , ^{99}Tc , and isotopes of thorium do not have a significant impact on the results of the dose analyses, either because inhalation pathways are relatively unimportant or because the inhalation dose conversion factors for other clearance classes are not significantly different.

Dose conversion factors for external exposure give effective dose-equivalent rates per unit concentration of radionuclides in specified source regions. These factors depend on the distribution of radionuclides in the source region, the amount of self-shielding provided by the source region, and the amount of shielding between the source region and the location of the exposed individual. Therefore, separate sets of external dose conversion factors are required for the assumed exposure pathways involving immersion in contaminated water while swimming, exposure to contaminated soil while gardening, and exposure to waste in disposal units during indoor residence or while excavating at the site.

For external exposure to radionuclides while swimming in contaminated water, the radionuclides are assumed to be uniformly distributed in a source region of infinite extent. The external dose-rate conversion factors for immersion in contaminated water, which have been adopted by Federal agencies for use in assessing radiation doses to the public (Eckerman and Ryman 1993), are given in Table G.4. For radioactive decay chains, the dose-rate factor for each shorter-lived decay product takes into account the branching fraction in the decay of its parent radionuclide given in Table G.1.

Table G.2. Internal dose conversion factors for ingestion of radionuclides^a

Nuclide ^b	f_1^c	Rem per μCi ingested	Nuclide ^b	f_1^c	Rem per μCi ingested
³ H	1.0	6.4×10^{-5}	¹⁵² Eu	0.001	6.5×10^{-3}
¹⁰ Be	0.005	4.7×10^{-3}	¹⁵⁴ Eu	0.001	9.5×10^{-3}
¹⁴ C	1.0	2.1×10^{-3}	¹⁵⁵ Eu	0.001	1.5×10^{-3}
²⁶ Al	0.01	1.5×10^{-2}	¹⁹⁴ Os	0.01	1.1×10^{-2}
³⁶ Cl	1.0	3.0×10^{-3}	²⁰⁷ Bi	0.05	5.5×10^{-3}
⁴⁰ K	1.0	1.9×10^{-2}	²¹⁰ Pb	0.2	5.4
⁴¹ Ca	0.3	1.3×10^{-3}	²¹⁰ Po	0.1	1.9
⁶⁰ Co	0.3	2.7×10^{-2}	²²⁶ Ra	0.2	1.3
⁵⁹ Ni	0.05	2.1×10^{-4}	²¹⁰ Pb ^d		
⁶³ Ni	0.05	5.8×10^{-4}	²²⁸ Ra	0.2	1.4
⁷⁹ Se	0.8	8.7×10^{-3}	²²⁸ Th	0.0002	4.0×10^{-1}
⁸⁷ Rb	1.0	4.9×10^{-3}	²²⁴ Ra	0.2	3.7×10^{-1}
⁹⁰ Sr	0.3	1.4×10^{-1}	²¹² Pb	0.2	4.6×10^{-2}
⁹⁰ Y	0.0001	1.1×10^{-2}	²²⁷ Ac	0.001	1.4×10^1
⁹³ Zr	0.002	1.7×10^{-3}	²²³ Ra	0.2	6.6×10^{-1}
^{93m} Nb	0.01	5.2×10^{-4}	²²⁹ Th	0.0002	3.5
⁹⁴ Nb	0.01	7.1×10^{-3}	²²⁵ Ra	0.2	3.8×10^{-1}
⁹⁹ Tc	0.8	1.5×10^{-3}	²²⁵ Ac	0.001	1.1×10^{-1}
¹⁰⁷ Pd	0.005	1.5×10^{-4}	²³⁰ Th	0.0002	5.5×10^{-1}
^{113m} Cd	0.05	1.6×10^{-1}	²³² Th	0.0002	2.7
^{121m} Sn	0.02	1.6×10^{-3}	²²⁸ Ra ^d		
¹²⁶ Sn	0.02	1.9×10^{-2}	²³¹ Pa	0.001	1.1×10^1
¹²⁶ Sb	0.1	1.0×10^{-2}	²²⁷ Ac ^d		
¹²⁹ I	1.0	2.8×10^{-1}	²³² U ^e	0.05	1.3
¹³⁵ Cs	1.0	7.1×10^{-3}	²³³ U	0.05	2.9×10^{-1}
¹³⁷ Cs	1.0	5.0×10^{-2}	²³⁴ U	0.05	2.8×10^{-1}
¹⁵¹ Sm	0.0003	3.9×10^{-4}	²³⁵ U	0.05	2.7×10^{-1}

Table G.2. (continued)^a

Nuclide ^b	f_1^c	Rem per μCi ingested	Nuclide ^b	f_1^c	Rem per μCi ingested
²³⁶ U	0.05	2.7×10^{-1}	^{242m} Am	0.001	3.5
²³⁸ U	0.05	2.5×10^{-1}	²⁴² Cm	0.001	1.1×10^{-1}
²³⁴ Th	0.0002	1.4×10^{-2}	²⁴³ Am	0.001	3.6
²³⁷ Np	0.001	4.4	²⁴³ Cm	0.001	2.5
²³⁸ Pu	0.001	3.2	²⁴⁴ Cm	0.001	2.0
²³⁹ Pu	0.001	3.5	²⁴⁵ Cm	0.001	3.7
²⁴⁰ Pu	0.001	3.5	²⁴⁶ Cm	0.001	3.7
²⁴¹ Pu	0.001	6.8×10^{-2}	²⁴⁷ Cm	0.001	3.4
²⁴² Pu	0.001	3.4	²⁴⁸ Cm	0.001	1.4×10^1
²⁴⁴ Pu	0.001	3.3	²⁴⁹ Cf	0.001	4.7
²⁴¹ Am	0.001	3.6	²⁵⁰ Cf	0.001	2.1
			²⁵¹ Cf	0.001	4.8

^aValues from Eckerman et al. (1988) give 50-year committed effective dose equivalent per unit activity ingested.

^bIndented entries are radiologically significant shorter-lived decay products of parent radionuclide listed.

^cFraction of ingested radionuclide absorbed into blood from gastrointestinal tract.

^dValues for radionuclide and its decay products are listed separately.

^eValues for shorter-lived decay products ²²⁸Th, ²²⁴Ra, and ²¹²Pb are listed following entry for ²²⁸Ra.

Table G.3. Internal dose conversion factors for inhalation of radionuclides^a

Nuclide ^b	Clearance class ^c	Rem per μCi inhaled	Nuclide ^b	Clearance class ^c	Rem per μCi inhaled
³ H ^d		9.6×10^{-5}	¹⁵² Eu	W	2.2×10^{-1}
¹⁰ Be	Y	3.5×10^{-1}	¹⁵⁴ Eu	W	2.9×10^{-1}
¹⁴ C ^e		2.1×10^{-3}	¹⁵⁵ Eu	W	4.1×10^{-2}
²⁶ Al	D	8.0×10^{-2}	¹⁹⁴ Os	Y	6.7×10^{-1}
³⁶ Cl	W	2.2×10^{-2}	²⁰⁷ Bi	W	2.0×10^{-2}
⁴⁰ K	D	1.2×10^{-2}	²¹⁰ Pb	D	1.4×10^1
⁴¹ Ca	W	1.3×10^{-3}	²¹⁰ Po	D	9.4
⁶⁰ Co	Y	2.2×10^{-1}	²²⁶ Ra	W	8.6
⁵⁹ Ni ^f	D	1.3×10^{-3}	²²² Rn ^g		
⁶³ Ni ^f	D	3.1×10^{-3}	²¹⁰ Pb ^h		
⁷⁹ Se	W	6.5×10^{-3}	²²⁸ Ra		—
⁸⁷ Rb	D	3.2×10^{-3}	²²⁸ Th	Y	3.4×10^2
⁹⁰ Sr	D	2.4×10^{-1}	²²⁰ Rn ^g		
⁹⁰ Y	Y	8.4×10^{-3}	²²⁷ Ac	D	6.7×10^3
⁹³ Zr	D	3.2×10^{-1}	²²⁹ Th	Y	1.7×10^3
^{93m} Nb	Y	2.9×10^{-2}	²³⁰ Th	Y	2.6×10^2
⁹⁴ Nb	Y	4.1×10^{-1}	²³² Th	Y	1.2×10^3
⁹⁹ Tc	D	1.0×10^{-3}	²²⁸ Ra ^h		
¹⁰⁷ Pd	Y	1.2×10^{-2}	²³¹ Pa	W	1.3×10^3
^{113m} Cd	D	1.5	²²⁷ Ac ^h		
^{121m} Sn	W	1.2×10^{-2}	²³² U ⁱ	Y	6.6×10^2
¹²⁶ Sn	W	1.0×10^{-1}	²³³ U	Y	1.4×10^2
¹²⁶ Sb	W	1.2×10^{-2}	²³⁴ U	Y	1.3×10^2
¹²⁹ I	D	1.7×10^{-1}	²³⁵ U	Y	1.2×10^2
¹³⁵ Cs	D	4.6×10^{-3}	²³⁶ U	Y	1.3×10^2
¹³⁷ Cs	D	3.2×10^{-2}	²³⁸ U	Y	1.2×10^2
¹⁵¹ Sm	W	3.0×10^{-2}	²³⁷ Np	W	5.4×10^2

Table G.3. (continued)^a

Nuclide ^b	Clearance class ^c	Rem per μCi inhaled	Nuclide ^b	Clearance class ^c	Rem per μCi inhaled
²³⁸ Pu	W	3.9×10^2	²⁴³ Am	W	4.4×10^2
²³⁹ Pu	W	4.3×10^2	²⁴³ Cm	W	3.1×10^2
²⁴⁰ Pu	W	4.3×10^2	²⁴⁴ Cm	W	2.5×10^2
²⁴¹ Pu	W	8.3	²⁴⁵ Cm	W	4.6×10^2
²⁴² Pu	W	4.1×10^2	²⁴⁶ Cm	W	4.5×10^2
²⁴⁴ Pu	W	4.0×10^2	²⁴⁷ Cm	W	4.1×10^2
²⁴¹ Am	W	4.4×10^2	²⁴⁸ Cm	W	1.7×10^3
^{242m} Am	W	4.3×10^2	²⁴⁹ Cf	W	5.8×10^2
²⁴² Cm	W	1.7×10^1	²⁵⁰ Cf	W	2.6×10^2
			²⁵¹ Cf	W	5.9×10^2

^aValues from Eckerman et al. (1988) give 50-year committed effective dose equivalent per unit activity inhaled.

^bIndented entries are radiologically significant shorter-lived decay products of parent radionuclide listed.

^cClearance from respiratory passages for radionuclides in particulate form in a matter of days (D), weeks (W), or years (Y).

^dValue applies to tritiated water and is increased by factor of 1.5 to include dose from absorption through the skin (ICRP 1979).

^eRadionuclide is assumed to be in organic form.

^fRadionuclide is assumed to be in inorganic form.

^gInhalation doses from isotopes of radon and their short-lived decay products are estimated using model described in Sect. G.5.2.1.

^hValues for radionuclide and its decay products are listed separately.

ⁱValues for shorter-lived decay products ²²⁸Th and ²²⁰Rn are listed following entry for ²²⁸Ra.

Table G.4. External dose-rate conversion factors for immersion
in contaminated water^d

Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci/L}$)
²⁶ Al	3.4×10^1	²²⁷ Ac	—
⁴⁰ K	2.0	²²⁷ Th	1.2
⁶⁰ Co	3.2×10^1	²²³ Ra	1.6
⁹⁴ Nb	2.0×10^1	²¹¹ Pb	6.3×10^{-1}
^{121m} Sn	1.6×10^{-2}	²¹¹ Bi	5.7×10^{-1}
¹²⁶ Sn	5.6×10^{-1}	²⁰⁷ Tl	3.9×10^{-2}
^{126m} Sb	1.9×10^1	²²⁹ Th	1.0
¹²⁶ Sb	4.9	²²⁵ Ra	7.6×10^{-2}
¹²⁹ I	1.0×10^{-1}	²²⁵ Ac	1.9×10^{-1}
¹³⁷ Cs	—	²²¹ Fr	3.8×10^{-1}
^{137m} Ba	6.9	²¹³ Bi	1.6
¹⁵² Eu	1.4×10^1	²⁰⁹ Tl	5.6×10^{-1}
¹⁵⁴ Eu	1.6×10^1	²³² Th	—
¹⁵⁵ Eu	6.6×10^{-1}	²²⁸ Ra ^c	—
²⁰⁷ Bi	1.9×10^1	²³¹ Pa	4.4×10^{-1}
²²⁶ Ra	—	²²⁷ Ac ^c	—
²¹⁴ Pb	3.0	²³² U ^d	—
²¹⁴ Bi	1.9×10^1	²³⁵ U	1.9
²²⁸ Ra	—	²³¹ Th	1.4×10^{-1}
²²⁸ Ac	1.2×10^1	²³⁸ U	—
²¹² Pb	1.8	²³⁴ Th	8.9×10^{-2}
²¹² Bi	2.3	^{234m} Pa	1.8×10^{-1}
²⁰⁸ Tl	1.6×10^1	²³⁴ Pa	3.8×10^{-2}

Table G.4. (continued)^a

Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci/L}$)
²³⁷ Np	2.7×10^{-1}	²⁴³ Am	5.8×10^{-1}
²³³ Pa	2.4	²³⁹ Np	2.0
²⁴⁴ Pu	—	²⁴³ Cm	1.5
^{240m} Np	4.1	²⁴⁵ Cm	1.0
²⁴¹ Am	2.2×10^{-1}	²⁴⁷ Cm	3.8
^{242m} Am	8.5×10^{-3}	²⁴³ Pu	2.7×10^{-1}
²⁴² Am	1.6×10^{-1}	²⁴⁹ Cf	4.0
²³⁸ Np	3.3×10^{-2}	²⁵¹ Cf	1.4

^aValues from Eckerman and Ryman (1993) give external effective dose-equivalent rate per unit activity concentration in water.

^bIndented entries are radiologically significant shorter-lived decay products of parent radionuclide listed; dose-rate factor for each decay product takes into account branching fraction in decay of parent radionuclide given in Table G.1.

^cValues for radionuclide and its decay products are listed separately.

^dValues for shorter-lived decay products ²¹²Pb, ²¹²Bi, and ²⁰⁸Tl are listed following entry for ²²⁸Ra.

For external exposure to radionuclides in contaminated soil while gardening or to radionuclides in disposal units, either during indoor residence or while excavating near disposal units, the source region is assumed to be a uniformly contaminated slab of infinite lateral extent. Depending upon the exposure scenario, as described below, the slab source is assumed to have either finite or infinite thickness, and the shielding provided by uncontaminated material between the source and receptor locations is taken into account. The idealized distributions of radionuclides in the source region assumed in the dose analysis probably are reasonable, because only about 1 m of soil-equivalent material between a slab source and a receptor location provides nearly complete shielding (Kocher and Sjoeren 1985). In all calculations of external dose from radionuclides in contaminated garden soil or in disposal units, an exposed individual is assumed to be located at a distance of 1 m from the source region. The small amount of shielding provided by 1 m of air can be neglected in all cases compared with the much greater shielding provided by the soil in the source region itself or by an engineered barrier.

For external exposure while working in the vegetable garden, the source region is assumed to be a slab extending from the ground surface to a depth of 15 cm, which is a typical thickness of a plowed layer. The external dose-rate conversion factors for this case are given in Table G.5. For external exposure while residing in the home on top of exposed waste, the source region is assumed

**Table G.5. External dose-rate conversion factors for radionuclides
uniformly distributed in 15 cm of surface soil^a**

Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci}/\text{m}^3$)
²⁶ Al	9.0×10^{-3}	²²⁷ Ac	—
⁴⁰ K	5.3×10^{-4}	²²⁷ Th	3.1×10^{-4}
⁶⁰ Co	8.5×10^{-3}	²²³ Ra	3.6×10^{-4}
⁹⁴ Nb	5.3×10^{-3}	²¹¹ Pb	1.7×10^{-4}
^{121m} Sn	1.2×10^{-6}	²¹¹ Bi	1.5×10^{-4}
¹²⁶ Sn	9.2×10^{-5}	²⁰⁷ Tl	1.1×10^{-5}
^{126m} Sb	5.2×10^{-3}	²²⁹ Th	2.0×10^{-4}
¹²⁶ Sb	1.3×10^{-3}	²²⁵ Ra	6.9×10^{-6}
¹²⁹ I	8.1×10^{-6}	²²⁵ Ac	3.9×10^{-5}
¹³⁷ Cs	—	²²¹ Fr	9.2×10^{-5}
^{137m} Ba	1.9×10^{-3}	²¹³ Bi	4.4×10^{-4}
¹⁵² Eu	3.8×10^{-3}	²⁰⁹ Tl	1.5×10^{-4}
¹⁵⁴ Eu	4.1×10^{-3}	²³² Th	—
¹⁵⁵ Eu	1.1×10^{-4}	²²⁸ Ra ^c	
²⁰⁷ Bi	5.1×10^{-3}	²³¹ Pa	1.1×10^{-4}
²²⁶ Ra	—	²²⁷ Ac ^c	
²¹⁴ Pb	7.8×10^{-4}	²³² U ^d	—
²¹⁴ Bi	5.1×10^{-3}	²³⁵ U	4.6×10^{-4}
²²⁸ Ra	—	²³¹ Th	2.3×10^{-5}
²²⁸ Ac	3.2×10^{-3}	²³⁸ U	—
²¹² Pb	4.2×10^{-4}	²³⁴ Th	1.5×10^{-5}
²¹² Bi	6.3×10^{-4}	^{234m} Pa	4.9×10^{-5}
²⁰⁸ Tl	4.1×10^{-3}	²³⁴ Pa	1.0×10^{-5}

Table G.5. (continued)^a

Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci}/\text{m}^3$)
²³⁷ Np	4.9×10^{-5}	²⁴³ Am	8.9×10^{-5}
²³³ Pa	6.0×10^{-4}	²³⁹ Np	4.6×10^{-4}
²⁴⁴ Pu	—	²⁴³ Cm	3.5×10^{-4}
^{240m} Np	1.1×10^{-3}	²⁴⁵ Cm	2.1×10^{-4}
²⁴¹ Am	2.7×10^{-5}	²⁴⁷ Cm	1.0×10^{-3}
^{242m} Am	1.1×10^{-6}	²⁴³ Pu	4.9×10^{-5}
²⁴² Am	3.1×10^{-5}	²⁴⁹ Cf	1.1×10^{-3}
²³⁸ Np	8.8×10^{-6}	²⁵¹ Cf	3.2×10^{-4}

^aValues from Eckerman and Ryman (1993) give external effective dose-equivalent rate per unit activity concentration in soil at distance of 1 m from source region.

^bIndented entries are radiologically significant shorter-lived decay products of parent radionuclide listed; dose-rate factor for each decay product takes into account branching fraction in decay of parent radionuclide given in Table G.1.

^cValues for radionuclide and its decay products are listed separately.

^dValues for shorter-lived decay products ²¹²Pb, ²¹²Bi, and ²⁰⁸Tl are listed following entry for ²²⁸Ra.

to be a slab of essentially infinite thickness, with no shielding assumed between the source region and the receptor location other than that provided by the soil in the source region (shielding provided by the walls of the home during indoor residence is taken into account by a separate parameter in the exposure pathway analysis itself). The external dose-rate conversion factors for this case are given in the column in Table G.6 labeled "No shielding." In both of these cases, the external dose-rate conversion factors have been adopted by Federal agencies for use in assessing radiation doses to the public (Eckerman and Ryman 1993). For all radioactive decay chains, the dose-rate factor for each short-lived decay product takes into account the branching fraction in the decay of its parent radionuclide given in Table G.1.

For external exposure in the resident and discovery scenarios, the source region also is assumed to be a slab of infinite thickness, but a thickness of uncontaminated shielding material between the source and receptor locations, equivalent to 15 or 30 cm of soil depending upon the particular disposal unit, is assumed to be provided by the engineered barrier used in constructing the units. The external dose-rate conversion factors for these two thicknesses of shielding are given in the last two columns in Table G.6. Since values for these cases were not given by Eckerman and Ryman (1993), the dose-rate factors in Table G.6 were obtained from calculated absorbed dose rates in air for monoenergetic photon sources in soil (Kocher and Sjoeren 1985) and the spectrum of

**Table G.6. External dose-rate conversion factors for radionuclides
uniformly distributed in infinite thickness of soil^a**

Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci}/\text{m}^3$)		
	No shielding ^c	15-cm shielding ^d	30-cm shielding ^d
²⁶ Al	1.1×10^{-2}	3.0×10^{-3}	9.4×10^{-4}
⁴⁰ K	6.5×10^{-4}	1.9×10^{-4}	5.9×10^{-5}
⁶⁰ Co	1.0×10^{-2}	2.9×10^{-3}	8.2×10^{-4}
⁹⁴ Nb	6.1×10^{-3}	1.4×10^{-3}	3.1×10^{-4}
^{121m} Sn	1.2×10^{-6}	—	—
¹²⁶ Sn	9.2×10^{-5}	—	—
^{126m} Sb	5.8×10^{-3}	1.2×10^{-3}	2.4×10^{-4}
¹²⁶ Sb	1.5×10^{-3}	3.1×10^{-4}	6.3×10^{-5}
¹²⁹ I	8.1×10^{-6}	—	—
¹³⁷ Cs	—	—	—
^{137m} Ba	2.1×10^{-3}	4.5×10^{-4}	9.0×10^{-5}
¹⁵² Eu	4.4×10^{-3}	1.1×10^{-3}	2.8×10^{-4}
¹⁵⁴ Eu	4.8×10^{-3}	1.2×10^{-3}	3.2×10^{-4}
¹⁵⁵ Eu	1.1×10^{-4}	2.0×10^{-6}	5.7×10^{-8}
²⁰⁷ Pb	5.9×10^{-3}	1.4×10^{-3}	3.5×10^{-4}
²²⁶ Ra	—	—	—
²¹⁴ Pb	8.4×10^{-4}	1.1×10^{-4}	1.6×10^{-5}
²¹⁴ Bi	6.1×10^{-3}	1.7×10^{-3}	5.2×10^{-4}
²²⁸ Ra	—	—	—
²²⁸ Ac	3.7×10^{-3}	8.7×10^{-4}	2.2×10^{-4}
²¹² Pb	4.4×10^{-4}	—	—
²¹² Bi	7.3×10^{-4}	1.8×10^{-4}	4.8×10^{-5}
²⁰⁸ Tl	5.2×10^{-3}	1.6×10^{-3}	6.0×10^{-4}

Table G.6. (continued)^a

Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci}/\text{m}^3$)		
	No shielding ^c	15-cm shielding ^d	30-cm shielding ^d
²²⁷ Ac	—	—	—
²²⁷ Th	3.2×10^{-4}	3.0×10^{-5}	3.2×10^{-6}
²²³ Ra	3.8×10^{-4}	3.3×10^{-5}	3.8×10^{-6}
²¹¹ Pb	1.9×10^{-4}	3.9×10^{-5}	8.0×10^{-6}
²¹¹ Bi	1.6×10^{-4}	2.3×10^{-5}	3.2×10^{-6}
²⁰⁷ Tl	1.2×10^{-5}	2.1×10^{-6}	5.0×10^{-7}
²²⁹ Th	2.0×10^{-4}	6.6×10^{-6}	3.7×10^{-7}
²²⁵ Ra	6.9×10^{-6}	—	—
²²⁵ Ac	4.0×10^{-5}	1.5×10^{-6}	1.1×10^{-7}
²²¹ Fr	9.6×10^{-5}	6.3×10^{-6}	4.9×10^{-7}
²¹³ Bi	4.8×10^{-4}	8.3×10^{-5}	1.4×10^{-5}
²⁰⁹ Tl	1.7×10^{-4}	4.9×10^{-5}	1.5×10^{-5}
²³² Th	—	—	—
²²⁸ Ra ^e	—	—	—
²³¹ Pa	1.2×10^{-4}	1.1×10^{-5}	1.3×10^{-6}
²²⁷ Ac ^e	—	—	—
²³² U ^f	—	—	—
²³⁵ U	4.5×10^{-4}	2.5×10^{-5}	1.6×10^{-6}
²³¹ Th	2.3×10^{-5}	—	—
²³⁸ U	—	—	—
²³⁴ Th	1.5×10^{-5}	—	—
^{234m} Pa	5.6×10^{-5}	1.1×10^{-5}	2.7×10^{-6}
²³⁴ Pa	1.2×10^{-5}	2.8×10^{-6}	6.8×10^{-7}
²³⁷ Np	4.9×10^{-5}	1.1×10^{-6}	5.1×10^{-8}
²³³ Pa	6.4×10^{-4}	8.0×10^{-5}	9.9×10^{-6}

Table G.6. (continued)^a

Nuclide ^b	Dose-rate factor (rem/year per $\mu\text{Ci}/\text{m}^3$)		
	No shielding ^c	15-cm shielding ^d	30-cm shielding ^d
²⁴⁴ Pu	—	—	—
^{240m} Np	1.3×10^{-3}	2.7×10^{-4}	6.0×10^{-5}
²⁴¹ Am	2.7×10^{-5}	3.6×10^{-8}	—
^{242m} Am	1.1×10^{-6}	1.2×10^{-8}	—
²⁴² Am	3.1×10^{-5}	7.9×10^{-7}	2.8×10^{-8}
²³⁸ Np	1.0×10^{-5}	2.6×10^{-6}	6.7×10^{-7}
²⁴³ Am	8.9×10^{-5}	6.0×10^{-7}	—
²³⁹ Np	4.7×10^{-4}	3.3×10^{-5}	3.0×10^{-6}
²⁴³ Cm	3.6×10^{-4}	2.5×10^{-5}	2.2×10^{-6}
²⁴⁵ Cm	3.6×10^{-4}	5.6×10^{-6}	2.8×10^{-7}
²⁴⁷ Cm	1.1×10^{-3}	1.7×10^{-4}	2.4×10^{-5}
²⁴³ Pu	5.0×10^{-5}	—	—
²⁴⁹ Cf	1.2×10^{-3}	1.7×10^{-4}	2.3×10^{-5}
²⁵¹ Cf	3.3×10^{-4}	1.6×10^{-5}	1.1×10^{-6}

^aValues from indicated sources give external effective dose-equivalent rate per unit activity concentration in soil at distance of 1 m from source region and assumed thicknesses of shielding by soil-equivalent material between source and receptor locations.

^bIndented entries are radiologically significant shorter-lived decay products of parent radionuclide listed.

^cValues from Eckerman and Ryman (1993).

^dValues based on calculations of absorbed dose in air from monoenergetic sources in soil by Kocher and Sjoeren (1985), spectrum of photons in decay of each radionuclide from Kocher (1981), and assumed factor for converting absorbed dose in air to effective dose equivalent of 0.8.

^eValues for radionuclide and its decay products are listed separately.

^fValues for shorter-lived decay products ²¹²Pb, ²¹²Bi, and ²⁰⁸Tl are listed following entry for ²²⁸Ra.

photons emitted by each radionuclide (Kocher 1981). The calculated absorbed dose in air was converted to effective dose equivalent in an exposed individual by multiplying by a factor of 0.8. This is an excellent approximation for all photon energies above about 0.1 MeV and provides a conservative estimate of dose for lower photon energies. For the exposure pathways represented in Table G.5 and the second column (No shielding) in Table G.6, the calculations from Kocher and Sjoeren (1985) agree with those from Eckerman and Ryman (1993) within a few tens of percent, but

generally are higher. Thus, the dose-rate factors in the last two columns of Table G.6 should be somewhat conservative. The dose-rate factor for short-lived radionuclides in decay chains again takes into account the branching fraction in the decay of the parent radionuclide given in Table G.1.

G.5 MODELS AND PARAMETER VALUES FOR EXPOSURE PATHWAYS

This section presents the models used to calculate doses to off-site individuals from use of contaminated water and doses to inadvertent intruders from direct intrusion into disposal units for the various exposure pathways described in Sects. G.3.1 and G.3.2, respectively. For each exposure pathway, the parameter values assumed in implementing the models are presented, and the results of the dose analyses are presented in summary tables which give effective dose equivalents per unit concentration of radionuclides in water or in disposal units at the time exposures are assumed to occur.

The parameter values selected for use in the models usually are intended to represent reasonable best estimates rather than worst-case values. This approach is appropriate when the hypothetical and conservative nature of the assumed exposure scenarios is considered, particularly the assumptions that exposures occur only at the locations of greatest estimated radionuclide concentrations in the environment or that inadvertent intrusion into the disposal facility will occur as postulated.

G.5.1 Exposure Pathways for Off-Site Individuals from Use of Contaminated Water

Intakes of contaminated groundwater or surface water by off-site individuals are assumed to result from the use of a contaminated source as a drinking water supply and from the ingestion of milk and meat obtained from dairy and beef cattle that drink contaminated water from the same source. Off-site individuals also are assumed to receive internal exposures from the consumption of contaminated fish obtained from surface water and to receive external exposures while swimming in contaminated surface water. As noted previously, exposures to contaminated groundwater or surface water are not considered in the dose analysis for inadvertent intruders (DOE 1996).

Drinking Water Pathway. The annual committed effective dose equivalent (rem per year) from direct ingestion of radionuclide i in drinking water (w) is given by

$$H_{iw} = C_{iw} U_w D_i, \quad (G.1)$$

where

- C_{iw} = concentration of radionuclide i in drinking water ($\mu\text{Ci/L}$),
- U_w = annual consumption of drinking water (L/year),
- D_i = dose conversion factor for ingestion of radionuclide i ($\text{rem}/\mu\text{Ci ingested}$).

In implementing the model, a daily consumption of contaminated drinking water of 2 L (i.e., an annual consumption of 730 L) is assumed (EPA 1989).

The model for estimating dose from the drinking water pathway is summarized in Table G.7. The annual dose per unit concentration of a radionuclide in water is obtained by multiplying the assumed annual consumption of drinking water by the ingestion dose conversion factor given in Table G.2.

Milk and Meat Pathways. The annual committed effective dose equivalents (rem per year) from ingestion of radionuclide i in milk (m) and meat (f) are given by

$$H_{im} = C_{im} U_m D_i \quad (G.2)$$

and

$$H_{if} = C_{if} U_f D_i, \quad (G.3)$$

respectively, where

- C_{im} = concentration of radionuclide i in milk ($\mu\text{Ci/L}$),
- C_{if} = concentration of radionuclide i in meat ($\mu\text{Ci/kg}$),
- U_m = annual consumption of milk (L/year),
- U_f = annual consumption of meat (kg/year),
- D_i = dose conversion factor for ingestion of radionuclide i (rem/ μCi ingested).

The dairy and beef cattle are assumed to drink only contaminated water, and the radionuclide concentrations in milk and meat are given by

$$C_{im} = C_{iw} Q_{wm} F_{im} \quad (G.4)$$

and

$$C_{if} = C_{iw} Q_{wf} F_{if}, \quad (G.5)$$

respectively, where

- C_{iw} = concentration of radionuclide i in drinking water ($\mu\text{Ci/L}$),
- Q_{wm} = daily consumption of drinking water by dairy cattle (L/d),
- Q_{wf} = daily consumption of drinking water by beef cattle (L/d),
- F_{im} = ratio of equilibrium concentration of radionuclide i in milk to daily intake by dairy cattle ($\mu\text{Ci/L}$ in milk per $\mu\text{Ci/d}$ intake),
- F_{if} = ratio of equilibrium concentration of radionuclide i in meat to daily intake by beef cattle ($\mu\text{Ci/kg}$ in meat per $\mu\text{Ci/d}$ intake).

**Table G.7. Annual effective dose equivalents from drinking water pathway
per unit concentration of radionuclides in water^a**

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
³ H	4.7×10^{-2}	¹⁵¹ Sm	2.8×10^{-1}
¹⁰ Be	3.4	¹⁵² Eu	4.7
¹⁴ C	1.5	¹⁵⁴ Eu	6.9
²⁶ Al	1.1×10^1	¹⁵⁵ Eu	1.1
³⁶ Cl	2.2	¹⁹⁴ Os	8.0
⁴⁰ K	1.4×10^1	²⁰⁷ Pb	4.0
⁴¹ Ca	9.5×10^{-1}	²¹⁰ Pb + d	5.3×10^3
⁶⁰ Co	2.0×10^1	²²⁶ Ra ^d	9.5×10^2
⁵⁹ Ni	1.5×10^{-1}	²²⁸ Ra + d	1.6×10^3
⁶³ Ni	4.2×10^{-1}	²²⁷ Ac + d	1.1×10^4
⁷⁹ Se	6.4	²²⁹ Th + d	2.9×10^3
⁸⁷ Rb	3.6	²³⁰ Th	4.0×10^2
⁹⁰ Sr + d	1.1×10^2	²³² Th ^e	2.0×10^3
⁹³ Zr ^c	1.2	²³¹ Pa ^f	8.0×10^3
^{93m} Nb	3.8×10^{-1}	²³² U + d	1.5×10^3
⁹⁴ Nb	5.2	²³³ U	2.1×10^2
⁹⁹ Tc	1.1	²³⁴ U	2.0×10^2
¹⁰⁷ Pd	1.1×10^{-1}	²³⁵ U	2.0×10^2
^{113m} Cd	1.2×10^2	²³⁶ U	2.0×10^2
^{121m} Sn	1.2	²³⁸ U + d	1.9×10^2
¹²⁶ Sn + d	1.5×10^1	²³⁷ Np	3.2×10^3
¹²⁹ I	2.0×10^2	²³⁸ Pu	2.3×10^3
¹³⁵ Cs	5.2	²³⁹ Pu	2.6×10^3
¹³⁷ Cs	3.7×10^1	²⁴⁰ Pu	2.6×10^3

Table G.7. (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
²⁴¹ Pu	5.0×10^1	²⁴⁵ Cm	2.7×10^3
²⁴² Pu	2.5×10^3	²⁴⁶ Cm	2.7×10^3
²⁴⁴ Pu	2.4×10^3	²⁴⁷ Cm	2.5×10^3
²⁴¹ Am	2.6×10^3	²⁴⁸ Cm	1.0×10^4
^{242m} Am + d ^g	2.6×10^3	²⁴⁹ Cf	3.4×10^3
²⁴³ Am	2.6×10^3	²⁵⁰ Cf	1.5×10^3
²⁴³ Cm	1.8×10^3	²⁵¹ Cf	3.5×10^3
²⁴⁴ Cm	1.5×10^3		

^aValues give 50-year committed effective dose equivalent from one-year intake of drinking water.

^b"m" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^ePossible contributions to dose from ²²⁸Ra decay product are not included.

^fPossible contributions to dose from ²²⁷Ac decay product are not included.

^gPossible contributions to dose from ²³⁸Pu decay product are not included.

In implementing the models, the assumed intake-to-milk transfer coefficients for dairy cattle (F_m) and intake-to-meat transfer coefficients for beef cattle (F_f) are given in Tables G.8 and G.9, respectively. The other parameter values assumed in the models are as follows: a daily consumption of water for dairy and beef cattle of 60 L and 50 L, respectively, and an annual consumption of milk and meat by an average adult of 110 L and 90 kg, respectively (NRC 1977).

The models for estimating dose from the milk and meat pathways are summarized in Tables G.10 and G.11, respectively. The annual dose per unit concentration of a radionuclide in water for each pathway is based on the models and parameter values described above and the ingestion dose conversion factor given in Table G.2.

Fish Pathway. The annual committed effective dose equivalent (rem per year) from the ingestion of radionuclide i in fish (q) is given by

$$H_{iq} = C_{iq} U_q D_i, \quad (\text{G.6})$$

where

- C_{iq} = concentration of radionuclide i in fish ($\mu\text{Ci/kg}$),
 U_q = annual consumption of fish (kg/year),
 D_i = dose conversion factor for ingestion of radionuclide i ($\text{rem}/\mu\text{Ci ingested}$).

Table G.8. Elemental intake-to-milk transfer coefficients for dairy cattle

Element	$F_m(\text{d/L})^a$	Element	$F_m(\text{d/L})^a$
H	1.4×10^{-2b}	Sb	1.0×10^{-4}
Be	9.0×10^{-7}	I	1.0×10^{-1}
C	1.5×10^{-2b}	Cs	7.0×10^{-3}
Al	2.0×10^{-4}	Sm	2.0×10^{-5}
Cl	1.5×10^{-2}	Eu	2.0×10^{-5}
K	7.0×10^{-3}	Os	5.0×10^{-3}
Ca	1.0×10^{-2}	Pb	2.5×10^{-4}
Co	2.0×10^{-3}	Bi	5.0×10^{-4}
Ni	1.0×10^{-3}	Po	3.5×10^{-4}
Se	4.0×10^{-3}	Ra	4.5×10^{-4}
Rb	1.0×10^{-2}	Ac	2.0×10^{-5}
Sr	1.5×10^{-3}	Th	5.0×10^{-6}
Y	2.0×10^{-5}	Pa	5.0×10^{-6}
Zr	3.0×10^{-5}	U	6.0×10^{-4}
Nb	2.0×10^{-2}	Np	5.0×10^{-6}
Tc	1.0×10^{-2}	Pu	1.0×10^{-7}
Pd	1.0×10^{-2}	Am	4.0×10^{-7}
Cd	1.0×10^{-3}	Cm	2.0×10^{-5}
Sn	1.0×10^{-3}	Cf	2.0×10^{-5c}

^aValues from Fig. 2.24 of Baes et al. (1984), except as noted.

^bValue from Table 4 of Ng (1982).

^cValue from Table 7 of Ng et al. (1977).

**Table G.9. Elemental intake-to-meat transfer
coefficients for beef cattle**

Element	$F(d/kg)^a$	Element	$F(d/kg)^a$
H	$1.2 \times 10^{-2}{}^b$	Sb	1.0×10^{-3}
Be	1.0×10^{-3}	I	7.0×10^{-3}
C	$3.1 \times 10^{-2}{}^b$	Cs	2.0×10^{-2}
Al	1.5×10^{-3}	Sm	5.0×10^{-3}
Cl	8.0×10^{-2}	Eu	5.0×10^{-3}
K	2.0×10^{-2}	Os	4.0×10^{-1}
Ca	7.0×10^{-4}	Pb	3.0×10^{-4}
Co	2.0×10^{-2}	Bi	4.0×10^{-4}
Ni	6.0×10^{-3}	Po	9.5×10^{-5}
Se	1.5×10^{-2}	Ra	2.5×10^{-4}
Rb	1.5×10^{-2}	Ac	2.5×10^{-5}
Sr	3.0×10^{-4}	Th	6.0×10^{-6}
Y	3.0×10^{-4}	Pa	1.0×10^{-5}
Zr	5.5×10^{-3}	U	2.0×10^{-4}
Nb	2.5×10^{-1}	Np	5.5×10^{-5}
Tc	8.5×10^{-3}	Pu	5.0×10^{-7}
Pd	4.0×10^{-3}	Am	3.5×10^{-6}
Cd	5.5×10^{-4}	Cm	3.5×10^{-6}
Sn	8.0×10^{-2}	Cf	$3.6 \times 10^{-6}{}^c$

^aValues from Fig. 2.25 of Baes et al. (1984), except as noted.

^bValue from Table E-1 of NRC (1977).

^cValue from Table 5.37 of Peterson (1983).

**Table G.10. Annual effective dose equivalents from milk pathway
per unit concentration of radionuclides in water^a**

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
³ H	5.9×10^{-3}	¹⁵¹ Sm	5.1×10^{-5}
¹⁰ Be	2.8×10^{-5}	¹⁵² Eu	8.6×10^{-4}
¹⁴ C	2.1×10^{-1}	¹⁵⁴ Eu	1.3×10^{-3}
²⁶ Al	2.0×10^{-2}	¹⁵⁵ Eu	2.0×10^{-4}
³⁶ Cl	3.0×10^{-1}	¹⁹⁴ Os	3.6×10^{-1}
⁴⁰ K	8.8×10^{-1}	²⁰⁷ Pb	1.5×10^{-2}
⁴¹ Ca	8.6×10^{-2}	²¹⁰ Pb + d	1.3×10^1
⁶⁰ Co	3.6×10^{-1}	²²⁶ Ra ^d	3.9
⁵⁹ Ni	1.4×10^{-3}	²²⁸ Ra + d	5.4
⁶³ Ni	3.8×10^{-3}	²²⁷ Ac + d	3.8
⁷⁹ Se	2.3×10^{-1}	²²⁹ Th + d	1.2
⁸⁷ Rb	3.2×10^{-1}	²³⁰ Th	1.8×10^{-2}
⁹⁰ Sr	1.4	²³² Th ^e	8.9×10^{-2}
⁹³ Zr ^c	3.4×10^{-4}	²³¹ Pa ^f	3.6×10^{-1}
^{93m} Nb	6.9×10^{-2}	²³² U + d	6.3
⁹⁴ Nb	9.4×10^{-1}	²³³ U	1.1
⁹⁹ Tc	9.9×10^{-2}	²³⁴ U	1.1
¹⁰⁷ Pd	9.9×10^{-3}	²³⁵ U	1.1
^{113m} Cd	1.1	²³⁶ U	1.1
^{121m} Sn	1.1×10^{-2}	²³⁸ U	1.0
¹²⁶ Sn	1.3×10^{-1}	²³⁷ Np	1.5×10^{-1}
¹²⁹ I	1.8×10^2	²³⁸ Pu	2.1×10^{-3}
¹³⁵ Cs	3.3×10^{-1}	²³⁹ Pu	2.3×10^{-3}
¹³⁷ Cs	2.3	²⁴⁰ Pu	2.3×10^{-3}

Table G.10. (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
²⁴¹ Pu	4.5×10^{-5}	²⁴⁵ Cm	4.9×10^{-1}
²⁴² Pu	2.2×10^{-3}	²⁴⁶ Cm	4.9×10^{-1}
²⁴⁴ Pu	2.2×10^{-3}	²⁴⁷ Cm	4.5×10^{-1}
²⁴¹ Am	9.5×10^{-3}	²⁴⁸ Cm	1.8
^{242m} Am + d ^g	2.1×10^{-2}	²⁴⁹ Cf	6.2×10^{-1}
²⁴³ Am	9.5×10^{-3}	²⁵⁰ Cf	2.8×10^{-1}
²⁴³ Cm	3.3×10^{-1}	²⁵¹ Cf	6.3×10^{-3}
²⁴⁴ Cm	2.6×10^{-1}		

^aValues give 50-year committed effective dose equivalent from one-year intake of milk.

^b" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^ePossible contributions to dose from ²²⁸Ra decay product are not included.

^fPossible contributions to dose from ²²⁷Ac decay product are not included.

^gPossible contributions to dose from ²³⁸Pu decay product are not included.

**Table G.11. Annual effective dose equivalents from meat pathway
per unit concentration of radionuclides in water^a**

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
³ H	3.5×10^{-3}	¹⁵¹ Sm	8.8×10^{-3}
¹⁰ Be	2.1×10^{-7}	¹⁵² Eu	1.5×10^{-1}
¹⁴ C	2.9×10^{-1}	¹⁵⁴ Eu	2.1×10^{-1}
²⁶ Al	1.0×10^{-1}	¹⁵⁵ Eu	3.4×10^{-2}
³⁶ Cl	1.1	¹⁹⁴ Os	2.0×10^1
⁴⁰ K	1.7	²⁰⁷ Pb	9.9×10^{-3}
⁴¹ Ca	4.1×10^{-3}	²¹⁰ Pb + d	8.1
⁶⁰ Co	2.4	²²⁶ Ra ^d	1.5
⁵⁹ Ni	5.7×10^{-3}	²²⁸ Ra + d	2.1
⁶³ Ni	1.6×10^{-2}	²²⁷ Ac + d	2.3
⁷⁹ Se	5.9×10^{-1}	²²⁹ Th + d	5.4×10^{-1}
⁸⁷ Rb	3.3×10^{-1}	²³⁰ Th	1.5×10^{-2}
⁹⁰ Sr + d	2.1×10^{-1}	²³² Th ^e	7.3×10^{-2}
⁹³ Zr ^c	4.2×10^{-2}	²³¹ Pa ^f	5.0×10^{-1}
^{93m} Nb	5.9×10^{-1}	²³² U + d	1.7
⁹⁴ Nb	8.0	²³³ U	2.6×10^{-1}
⁹⁹ Tc	5.7×10^{-2}	²³⁴ U	2.5×10^{-1}
¹⁰⁷ Pd	2.7×10^{-3}	²³⁵ U	2.4×10^{-1}
^{113m} Cd	4.0×10^{-1}	²³⁶ U	2.4×10^{-1}
^{121m} Sn	5.8×10^{-1}	²³⁸ U	2.3×10^{-1}
¹²⁶ Sn	6.8	²³⁷ Np	1.1
¹²⁹ I	8.8	²³⁸ Pu	7.2×10^{-3}
¹³⁵ Cs	6.4×10^{-1}	²³⁹ Pu	7.9×10^{-3}
¹³⁷ Cs	4.5	²⁴⁰ Pu	7.9×10^{-3}

Table G.11. (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
²⁴¹ Pu	1.5×10^{-4}	²⁴⁵ Cm	5.8×10^{-2}
²⁴² Pu	7.7×10^{-3}	²⁴⁶ Cm	5.8×10^{-2}
²⁴⁴ Pu	7.4×10^{-3}	²⁴⁷ Cm	5.4×10^{-2}
²⁴¹ Am	5.7×10^{-2}	²⁴⁸ Cm	2.2×10^{-1}
^{242m} Am + d ^g	5.6×10^{-2}	²⁴⁹ Cf	7.6×10^{-2}
²⁴³ Am	5.7×10^{-2}	²⁵⁰ Cf	3.4×10^{-2}
²⁴³ Cm	3.9×10^{-2}	²⁵¹ Cf	7.8×10^{-2}
²⁴⁴ Cm	3.2×10^{-2}		

^aValues give 50-year committed effective dose equivalent from one-year intake of meat.

^b"n + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^ePossible contributions to dose from ²²⁸Ra decay product are not included.

^fPossible contributions to dose from ²²⁷Ac decay product are not included.

^gPossible contributions to dose from ²³⁸Pu decay product are not included.

Radionuclide concentrations in fish are given by

$$C_{iq} = B_{iq} C_{iw} / \rho_w, \quad (\text{G.7})$$

where

B_{iq} = fish-to-water concentration ratio for radionuclide i ($\mu\text{Ci/kg}$ wet weight in fish per $\mu\text{Ci/L}$ in water),

C_{iw} = concentration of radionuclide i in water ($\mu\text{Ci/L}$),

ρ_w = density of water (kg/L).

In implementing the model, the assumed fish-to-water concentration ratios are given in Table G.12. These values, which apply to freshwater fish, were adopted for use in a generic dose assessment for the NRC (Kennedy and Strenge 1992), based on a review of published values. The

Table G.12. Elemental fish-to-water concentration ratios for freshwater fish

Element	$B_q(\text{L/kg})^a$	Element	$B_q(\text{L/kg})^a$
H	1.0	Sb	2.0×10^2
Be	2.0	I	5.0×10^2
C	4.6×10^3	Cs	2.0×10^3
Al	$1.0 \times 10^2{}^b$	Sm	2.5×10^1
Cl	5.0×10^1	Eu	2.5×10^1
K	1.0×10^3	Os	1.0×10^1
Ca	4.0×10^1	Pb	1.0×10^2
Co	3.3×10^2	Bi	1.5×10^1
Ni	1.0×10^2	Po	5.0×10^2
Se	1.7×10^2	Ra	7.0×10^1
Rb	2.0×10^3	Ac	2.5×10^1
Sr	5.0×10^1	Th	1.0×10^2
Y	2.5×10^1	Pa	1.1×10^1
Zr	2.0×10^2	U	5.0×10^1
Nb	2.0×10^2	Np	2.5×10^2
Tc	1.5×10^1	Pu	2.5×10^2
Pd	1.0×10^1	Am	2.5×10^2
Cd	2.0×10^2	Cm	2.5×10^2
Sn	3.0×10^3	Cf	2.5×10^1

^aValues from Table 6.19 of Kennedy and Strenge (1992), except as noted.

^bValue from Table 6 of Chapman et al. (1968).

annual consumption of fish by an average adult was assumed to be 7 kg (NRC 1977), and the density of water is 1 kg/L.

The model for estimating dose from the fish pathway is summarized in Table G.13. The annual dose per unit concentration of a radionuclide in water is based on the model and parameter values described above and the ingestion dose conversion factor given in Table G.2.

External Exposure Pathway. For external exposure (e) while swimming in contaminated surface water, the annual effective dose equivalent (rem per year) from radionuclide i is given by

$$H_{ie} = C_{iw} U_w D_{iw}, \quad (\text{G.8})$$

where

- C_{iw} = concentration of radionuclide i in water ($\mu\text{Ci/L}$),
- U_w = fraction of the year during which external exposure to contaminated water while swimming occurs,
- D_{iw} = dose conversion factor for external exposure to radionuclide i in water (rem/year per $\mu\text{Ci/L}$).

In implementing the model, a fraction of the year during which exposure while swimming occurs of 0.01, which corresponds to an exposure time of about 100 hours per year, is assumed (DOE 1988b).

The model for estimating dose from external exposure while swimming in contaminated surface water is summarized in Table G.14. The annual dose per unit concentration of a radionuclide in water is obtained by multiplying the assumed fraction of the year during which exposure occurs by the external dose-rate conversion factor given in Table G.4.

All Exposure Pathways. The annual doses to off-site individuals per unit concentration of radionuclides in water from all exposure pathways combined are summarized in Table G.15 for contaminated groundwater and in Table G.16 for contaminated surface water. For exposure to contaminated groundwater, the dose in Table G.15 is the sum of the doses from the drinking water, milk, and meat pathways given in Tables G.7, G.10, and G.11, respectively.

For releases to surface water, off-site individuals could be exposed to contamination in a small surface stream (i.e., White Oak Creek) or in a large river (i.e., the Clinch River), and separate doses for these two cases are given in Table G.16. For exposure to contamination in the Clinch River, all exposure pathways would contribute to the dose, and the dose for this case in Table G.16 is the sum of the doses from the drinking water, milk, meat, fish, and swimming pathways given in Tables G.7, G.10, G.11, G.13, and G.14, respectively. For exposure to contamination in White Oak Creek, the fish pathway is assumed not to be relevant, either because fish consumed by off-site individuals would be obtained primarily from the Clinch River or because fish that might be obtained from White Oak Creek would spend most of their time in the Clinch River and, thus, the concentrations of radionuclides in the fish would be based primarily on the concentrations in the river. Therefore, the dose from exposure to contamination in White Oak Creek in Table G.16 is the sum of the doses from the drinking water, milk, meat, and swimming pathways only.

**Table G.13. Annual effective dose equivalents from fish pathway
per unit concentration of radionuclides in water^a**

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
³ H	4.5×10^{-4}	¹⁵¹ Sm	6.8×10^{-2}
¹⁰ Be	6.6×10^{-2}	¹⁵² Eu	1.1
¹⁴ C	6.8×10^1	¹⁵⁴ Eu	1.7
²⁶ Al	1.1×10^1	¹⁵⁵ Eu	2.6×10^{-1}
³⁶ Cl	1.1	¹⁹⁴ Os	7.7×10^{-1}
⁴⁰ K	1.3×10^2	²⁰⁷ Pb	5.8×10^{-1}
⁴¹ Ca	3.6×10^{-1}	²¹⁰ Pb + d	1.1×10^4
⁶⁰ Co	6.2×10^1	²²⁶ Ra ^d	6.4×10^2
⁵⁹ Ni	1.5×10^{-1}	²²⁸ Ra + d	1.2×10^3
⁶³ Ni	4.1×10^{-1}	²²⁷ Ac + d	2.8×10^3
⁷⁹ Se	1.0×10^1	²²⁹ Th + d	2.7×10^3
⁸⁷ Rb	6.9×10^1	²³⁰ Th	3.9×10^2
⁹⁰ Sr + d	5.1×10^1	²³² Th ^e	1.9×10^3
⁹³ Zr ^f	2.4	²³¹ Pa ^f	8.5×10^2
^{93m} Nb	7.3×10^{-1}	²³² U + d	9.5×10^2
⁹⁴ Nb	9.9	²³³ U	1.0×10^2
⁹⁹ Tc	1.6×10^{-1}	²³⁴ U	9.8×10^1
¹⁰⁷ Pd	1.1×10^{-2}	²³⁵ U	9.5×10^1
^{113m} Cd	2.2×10^2	²³⁶ U	9.5×10^1
^{121m} Sn	3.4×10^1	²³⁸ U + d	9.8×10^1
¹²⁶ Sn + d	4.1×10^2	²³⁷ Np	7.7×10^3
¹²⁹ I	9.8×10^2	²³⁸ Pu	5.6×10^3
¹³⁵ Cs	9.9×10^1	²³⁹ Pu	6.1×10^3
¹³⁷ Cs	7.0×10^2	²⁴⁰ Pu	6.1×10^3

Table G.13. (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
²⁴¹ Pu	1.2×10^2	²⁴⁵ Cm	6.5×10^3
²⁴² Pu	6.0×10^3	²⁴⁶ Cm	6.5×10^3
²⁴⁴ Pu	5.8×10^3	²⁴⁷ Cm	6.0×10^3
²⁴¹ Am	6.3×10^3	²⁴⁸ Cm	2.5×10^4
^{242m} Am + d ^g	6.3×10^3	²⁴⁹ Cf	8.2×10^2
²⁴³ Am	6.3×10^3	²⁵⁰ Cf	3.7×10^2
²⁴³ Cm	4.4×10^3	²⁵¹ Cf	8.4×10^2
²⁴⁴ Cm	3.5×10^3		

^aValues give 50-year committed effective dose equivalent from one-year intake of fish.

^b"m" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^ePossible contributions to dose from ²²⁸Ra decay product are not included.

^fPossible contributions to dose from ²²⁷Ac decay product are not included.

^gPossible contributions to dose from ²³⁸Pu decay product are not included.

However, off-site individuals who obtain their domestic water supply from White Oak Creek also are assumed to obtain fish from the Clinch River. Therefore, for individuals who are assumed to use water from White Oak Creek, the dose from all exposure pathways would be obtained as the sum of two terms: (1) the dose for White Oak Creek in Table G.16 applied to the radionuclide concentrations in the creek and (2) the dose for the fish pathway in Table G.13 applied to the radionuclide concentrations in the river. Because of the much higher flow rate in the Clinch River compared with White Oak Creek, the radionuclide concentrations in the river used in estimating the dose for the fish pathway would be substantially less than the concentrations in the creek used in estimating the dose for all the other exposure pathways.

Table G.14. Annual effective dose equivalents from external exposure while swimming per unit concentration of radionuclides in water

Nuclide ^a	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^a	Annual dose (rem/year per $\mu\text{Ci/L}$)
²⁶ Al	3.4×10^{-1}	²³² Th ^b	—
⁴⁰ K	2.0×10^{-2}	²³¹ Pa ^c	4.4×10^{-3}
⁶⁰ Co	3.2×10^{-1}	²³² U + d	2.0×10^{-1}
⁹⁴ Nb	2.0×10^{-1}	²³⁵ U + d	2.0×10^{-2}
^{121m} Sn	1.6×10^{-4}	²³⁸ U + d	3.1×10^{-3}
¹²⁶ Sn + d	2.4×10^{-1}	²³⁷ Np + d	2.7×10^{-2}
¹²⁹ I	1.0×10^{-3}	²⁴⁴ Pu + d	4.1×10^{-2}
¹³⁷ Cs + d	6.9×10^{-2}	²⁴¹ Am	2.2×10^{-3}
¹⁵² Eu	1.4×10^{-1}	^{242m} Am + d	2.0×10^{-3}
¹⁵⁴ Eu	1.6×10^{-1}	²⁴³ Am + d	2.6×10^{-2}
¹⁵⁵ Eu	6.6×10^{-3}	²⁴³ Cm	1.5×10^{-2}
²⁰⁷ Bi	1.9×10^{-1}	²⁴⁵ Cm	1.0×10^{-2}
²²⁶ Ra + d	2.2×10^{-1}	²⁴⁷ Cm + d	4.1×10^{-2}
²²⁸ Ra + d	3.2×10^{-1}	²⁴⁹ Cf	4.0×10^{-2}
²²⁷ Ac + d	4.0×10^{-2}	²⁵¹ Cf	1.4×10^{-2}
²²⁹ Th + d	3.8×10^{-2}		

^a" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^bPossible contributions to dose from ²²⁸Ra decay product are not included.

^cPossible contributions to dose from ²²⁷Ac decay product are not included.

Table G.15. Annual effective dose equivalents to off-site individuals per unit concentration of radionuclides in groundwater from all exposure pathways^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
³ H	5.6×10^{-2}	¹⁵¹ Sm	2.9×10^{-1}
¹⁰ Be	3.4	¹⁵² Eu	4.9
¹⁴ C	2.0	¹⁵⁴ Eu	7.1
²⁶ Al	1.1×10^1	¹⁵⁵ Eu	1.1
³⁶ Cl	3.6	¹⁹⁴ Os	2.8×10^1
⁴⁰ K	1.7×10^1	²⁰⁷ Pb	4.0
⁴¹ Ca	1.0	²¹⁰ Pb + d	5.3×10^3
⁶⁰ Co	2.3×10^1	²²⁶ Ra ^d	9.6×10^2
⁵⁹ Ni	1.7×10^{-1}	²²⁸ Ra + d	1.6×10^3
⁶³ Ni	4.4×10^{-1}	²²⁷ Ac + d	1.1×10^4
⁷⁹ Se	7.2	²²⁹ Th + d	2.9×10^3
⁸⁷ Rb	4.3	²³⁰ Th	4.0×10^2
⁹⁰ Sr + d	1.1×10^2	²³² Th ^e	2.0×10^3
⁹³ Zr ^c	1.2	²³¹ Pa ^f	8.0×10^3
^{93m} Nb	1.0	²³² U + d	1.5×10^3
⁹⁴ Nb	1.4×10^1	²³³ U	2.1×10^2
⁹⁹ Tc	1.3	²³⁴ U	2.0×10^2
¹⁰⁷ Pd	1.2×10^{-1}	²³⁵ U	2.0×10^2
^{113m} Cd	1.2×10^2	²³⁶ U	2.0×10^2
^{121m} Sn	1.8	²³⁸ U + d	1.9×10^2
¹²⁶ Sn + d	2.2×10^1	²³⁷ Np	3.2×10^3
¹²⁹ I	3.9×10^2	²³⁸ Pu	2.3×10^3
¹³⁵ Cs	6.2	²³⁹ Pu	2.6×10^3
¹³⁷ Cs	4.4×10^1	²⁴⁰ Pu	2.6×10^3

Table G.15. (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)
²⁴¹ Pu	5.0×10^1	²⁴⁵ Cm	2.7×10^3
²⁴² Pu	2.5×10^3	²⁴⁶ Cm	2.7×10^3
²⁴⁴ Pu	2.4×10^3	²⁴⁷ Cm	2.5×10^3
²⁴¹ Am	2.6×10^3	²⁴⁸ Cm	1.0×10^4
^{242m} Am + d ^g	2.6×10^3	²⁴⁹ Cf	3.4×10^3
²⁴³ Am	2.6×10^3	²⁵⁰ Cf	1.5×10^3
²⁴³ Cm	1.8×10^3	²⁵¹ Cf	3.5×10^3
²⁴⁴ Cm	1.5×10^3		

^aValues give sum of effective dose equivalents per unit concentration in water for drinking water, milk, and meat pathways given in Tables G.7, G.10, and G.11, respectively.

^b" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^ePossible contributions to dose from ²²⁸Ra decay product are not included.

^fPossible contributions to dose from ²²⁷Ac decay product are not included.

^gPossible contributions to dose from ²³⁸Pu decay product are not included.

Table G.16. Annual effective dose equivalents to off-site individuals per unit concentration of radionuclides in surface water from all exposure pathways^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)		Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	
	White Oak Creek ^c	Clinch River ^d		White Oak Creek ^c	Clinch River ^d
³ H	5.6×10^{-2}	5.6×10^{-2}	¹³⁵ Cs	6.2	1.1×10^2
¹⁰ Be	3.4	3.5	¹³⁷ Cs	4.4×10^1	7.4×10^2
¹⁴ C	2.0	7.0×10^1	¹⁵¹ Sm	2.9×10^{-1}	3.6×10^{-1}
²⁶ Al	1.1×10^1	2.2×10^1	¹⁵² Eu	5.0	6.1
³⁶ Cl	3.6	4.7	¹⁵⁴ Eu	7.3	9.0
⁴⁰ K	1.7×10^1	1.5×10^2	¹⁵⁵ Eu	1.1	1.4
⁴¹ Ca	1.0	1.4	¹⁹⁴ Os	2.8×10^1	2.9×10^1
⁶⁰ Co	2.3×10^1	8.5×10^1	²⁰⁷ Pb	4.2	4.8
⁵⁹ Ni	1.7×10^{-1}	3.2×10^{-1}	²¹⁰ Pb + d	5.3×10^3	1.6×10^4
⁶³ Ni	4.4×10^{-1}	8.5×10^{-1}	²²⁶ Ra ^f	9.6×10^2	1.6×10^3
⁷⁹ Se	7.2	1.7×10^1	²²⁸ Ra + d	1.6×10^3	2.8×10^3
⁸⁷ Rb	4.3	7.3×10^1	²²⁷ Ac + d	1.1×10^4	1.4×10^4
⁹⁰ Sr + d	1.1×10^2	1.6×10^2	²²⁹ Th + d	2.9×10^3	5.6×10^3
⁹³ Zr ^e	1.2	3.6	²³⁰ Th	4.0×10^2	7.9×10^2
^{93m} Nb	1.0	1.7	²³² Th ^g	2.0×10^3	3.9×10^3
⁹⁴ Nb	1.4×10^1	2.4×10^1	²³¹ Pa ^h	8.0×10^3	8.9×10^3
⁹⁹ Tc	1.3	1.5	²³² U + d	1.5×10^3	2.5×10^3
¹⁰⁷ Pd	1.2×10^{-1}	1.3×10^{-1}	²³³ U	2.1×10^2	3.1×10^2
^{113m} Cd	1.2×10^2	3.4×10^2	²³⁴ U	2.0×10^2	3.0×10^2
^{121m} Sn	1.8	3.6×10^1	²³⁵ U + d	2.0×10^2	3.0×10^2
¹²⁶ Sn + d	2.2×10^1	4.3×10^2	²³⁶ U	2.0×10^2	3.0×10^2
¹²⁹ I	3.9×10^2	1.4×10^3	²³⁸ U + d	1.9×10^2	2.9×10^2

Table G.16. (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)		Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/L}$)	
	White Oak Creek ^c	Clinch River ^d		White Oak Creek ^c	Clinch River ^d
²³⁷ Np	3.2×10^3	1.1×10^4	²⁴³ Cm	1.8×10^3	6.2×10^3
²³⁸ Pu	2.3×10^3	7.9×10^3	²⁴⁴ Cm	1.5×10^3	5.0×10^3
²³⁹ Pu	2.6×10^3	8.7×10^3	²⁴⁵ Cm	2.7×10^3	9.2×10^3
²⁴⁰ Pu	2.6×10^3	8.7×10^3	²⁴⁶ Cm	2.7×10^3	9.2×10^3
²⁴¹ Pu	5.0×10^1	1.7×10^2	²⁴⁷ Cm	2.5×10^3	8.5×10^3
²⁴² Pu	2.5×10^3	8.5×10^3	²⁴⁸ Cm	1.0×10^4	3.5×10^4
²⁴⁴ Pu	2.4×10^3	8.2×10^3	²⁴⁹ Cf	3.4×10^3	4.2×10^3
²⁴¹ Am	2.6×10^3	8.9×10^3	²⁵⁰ Cf	1.5×10^3	1.9×10^3
^{242m} Am + d ⁱ	2.6×10^3	8.9×10^3	²⁵¹ Cf	3.5×10^3	4.3×10^3
²⁴³ Am	2.6×10^3	8.9×10^3			

^aAssumed exposure pathways include drinking water, milk, meat, fish, and swimming; effective dose equivalents per unit concentration in water for these pathways are given in Tables G.7, G.10, G.11, G.13, and G.14, respectively.

^b"+ d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cDose from exposure to radionuclides in White Oak Creek is sum of doses from drinking water, milk, meat, and swimming pathways only. Off-site individuals who use White Oak Creek as domestic water supply also are assumed to obtain fish from Clinch River, and contribution to dose from fish pathway is based on doses per unit concentration of radionuclides in water in Table G.13 and radionuclide concentrations in Clinch River.

^dDose from exposure to radionuclides in Clinch River is sum of doses from all exposure pathways (i.e., drinking water, milk, meat, fish, and swimming).

^ePossible contributions to dose from ^{93m}Nb decay product are not included.

^fPossible contributions to dose from ²¹⁰Pb decay product are not included.

^gPossible contributions to dose from ²²⁸Ra decay product are not included.

^hPossible contributions to dose from ²²⁷Ac decay product are not included.

ⁱPossible contributions to dose from ²³⁸Pu decay product are not included.

G.5.2 Exposure Pathways for Direct Intrusion into Disposal Units

Exposures of inadvertent intruders resulting from direct intrusion into disposal units after loss of active institutional controls over the disposal site are assumed to occur according to the agriculture, resident, discovery, or post-drilling scenarios. This section presents the models and parameter values used to estimate effective dose equivalents per unit concentration of radionuclides in disposal units for the different exposure pathways assumed for each exposure scenario.

G.5.2.1 Agriculture Scenario

In the agriculture scenario, an inadvertent intruder is assumed to exhume waste materials from disposal units in digging a foundation for a home on top of the units, and some of the exhumed material is assumed to be mixed with native soil in the intruder's vegetable garden. The pathways for chronic exposure assumed for this scenario then include (1) ingestion of vegetables grown in contaminated soil, (2) direct ingestion of contaminated soil from the garden in conjunction with vegetable intakes, (3) external exposure to contaminated soil while working in the garden and during indoor residence on top of the disposal units, and (4) inhalation of activity suspended in air while working in the garden and during indoor residence.

Vegetable Pathway. The annual committed effective dose equivalent (rem per year) from ingestion of radionuclide i in vegetables (v) is given by

$$H_{iv} = C_{iv} U_v D_i, \quad (\text{G.9})$$

where

- C_{iv} = concentration of radionuclide i in vegetables ($\mu\text{Ci/kg}$),
- U_v = annual consumption of vegetables (kg/year),
- D_i = dose conversion factor for ingestion of radionuclide i ($\text{rem}/\mu\text{Ci ingested}$).

Radionuclides are assumed to be transferred to vegetables via root uptake from the contaminated garden soil. Radionuclide concentrations in vegetables then are given by

$$\begin{aligned} C_{iv} &= B_{iv} C_{is} / \rho_s \\ &= B_{iv} f_s C_{is} / \rho_s, \end{aligned} \quad (\text{G.10})$$

where

- B_{iv} = plant-to-soil concentration ratio for radionuclide i ($\mu\text{Ci/kg}$ wet weight in vegetation per $\mu\text{Ci/kg}$ dry weight in soil),
- C_{is} = concentration of radionuclide i in soil in vegetable garden ($\mu\text{Ci/m}^3$),

- ρ_s = density of soil (kg/m^3),
 C_i = concentration of radionuclide i in disposal units ($\mu\text{Ci/m}^3$),
 f_s = dilution factor for mixing of materials exhumed from disposal units into soil in vegetable garden.

In implementing the model, the assumed plant-to-soil concentration ratios in vegetables are given in Table G.17. Although limited site-specific data are available for a few radionuclides and plant types [e.g., see Garten et al. (1987)], site-specific data for the many radionuclides of concern to this analysis and for a variety of food crops are lacking. Therefore, except for the value for ^{14}C , which is based on a single set of measurements (Sheppard et al. 1991), and the value for ^3H , which is recommended by the NRC (1977), the plant-to-soil concentration ratios adopted in this analysis are generic, and were obtained from the compilation by Baes et al. (1984). Although this approach clearly is judgmental for application to the ORR, selection of the concentration ratios primarily from a single source at least ensures some degree of consistency among the values for the different elements, because similar biases presumably were applied by the evaluators in developing the adopted concentration ratios from the variety of published data.

Baes et al. (1984) give two sets of concentration ratios in vegetation—one for the vegetative portions of food crops, which would apply to leafy vegetables, and the other for the nonvegetative (reproductive) portions, which would apply to nonleafy vegetables. The values for nonvegetative portions of food crops were adopted for use in this analysis, because consumption of nonleafy vegetables generally is about an order of magnitude greater than consumption of leafy vegetables (Baes et al. 1984; Hamby 1992). In addition, the reported concentration ratios on a dry-weight basis for nonleafy vegetables were converted to a fresh-weight basis by multiplying by a factor of 0.43, which represents the average dry-weight-to-fresh-weight conversion factor for all types of nonleafy vegetables (Baes et al. 1984). Taking into account the concentration ratios for leafy vegetables would not significantly change the adopted values in Table G.17, because of the relatively low consumption of leafy vegetables and the much higher fresh-weight-to-dry-weight ratio for leafy vegetables than for nonleafy vegetables (Peterson 1983).

The other parameter values assumed in the model for the vegetable pathway are as follows: a dilution factor for mixing of exhumed materials, including both contaminated (i.e., waste) and uncontaminated materials, from the disposal units into native soil in the vegetable garden of 0.2 (Oztunali et al. 1981; NRC 1982; Napier et al. 1984); a density of soil of $1,400 \text{ kg/m}^3$ (Baes and Sharp 1983); and an annual consumption of contaminated vegetables, including both leafy and nonleafy vegetables, of 90 kg. The dilution factor for mixing exhumed materials with native soil in a vegetable garden is based on the reasonable assumption that no more than a relatively small fraction of the soil in the garden could be exhumed materials in order for the garden to be fertile, since essentially all exhumed materials would be obtained from well below the ground surface and, thus, would be nonfertile. The assumed annual consumption of leafy and nonleafy vegetables from the

Table G.17. Elemental plant-to-soil concentration ratios in vegetables

Element	B_v^a	Element	B_v^a
H	4.8 ^b	Sb	1.3×10^{-2}
Be	6.5×10^{-4}	I	2.2×10^{-2}
C	5.6×10^{-1c}	Cs	1.3×10^{-2}
Al	2.8×10^{-4}	Sm	1.7×10^{-3}
Cl	3.0×10^1	Eu	1.7×10^{-3}
K	2.4×10^{-1}	Os	1.5×10^{-3}
Ca	1.5×10^{-1}	Pb	3.9×10^{-3}
Co	3.0×10^{-3}	Bi	2.2×10^{-3}
Ni	2.6×10^{-2}	Po	1.7×10^{-4}
Se	1.1×10^{-2}	Ra	6.5×10^{-4}
Rb	3.0×10^{-2}	Ac	1.5×10^{-4}
Sr	1.1×10^{-1}	Th	3.7×10^{-5}
Y	2.6×10^{-3}	Pa	1.1×10^{-4}
Zr	2.2×10^{-4}	U	1.7×10^{-3}
Nb	2.2×10^{-3}	Np	4.3×10^{-3}
Tc	6.5×10^{-1}	Pu	1.9×10^{-5}
Pd	1.7×10^{-2}	Am	1.1×10^{-4}
Cd	6.5×10^{-2}	Cm	6.5×10^{-6}
Sn	2.6×10^{-3}	Cf	6.5×10^{-6d}

^aμCi/kg fresh weight of vegetation per μCi/kg dry weight of soil; except as noted, values are obtained from concentration ratios for dry weight of vegetation given in Fig. 2.2 of Baes et al. (1984) multiplied by factor of 0.43 to convert to fresh weight of vegetation (Baes et al. 1984).

^bValue from Table E-1 of NRC (1977).

^cValue from Sheppard et al. (1991) for carbonate form in acidic soil with low organic matter content; concentration ratio for dry weight of vegetation is multiplied by factor of 0.43 to convert to fresh weight of vegetation (Baes et al. 1984).

^dValue is assumed to be the same as that for Cm.

Table G.18. Annual effective dose equivalents to inadvertent intruders from vegetable pathway per unit concentration of radionuclides in disposal units for agriculture scenario^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
³ H	4.0×10^{-6}	¹³⁷ Cs	8.5×10^{-6}
¹⁰ Be	4.0×10^{-8}	¹⁵¹ Sm	8.6×10^{-9}
¹⁴ C	1.5×10^{-5}	¹⁵² Eu	1.4×10^{-7}
²⁶ Al	5.5×10^{-8}	¹⁵⁴ Eu	2.1×10^{-7}
³⁶ Cl	1.2×10^{-3}	¹⁵⁵ Eu	3.3×10^{-8}
⁴⁰ K	5.9×10^{-5}	¹⁹⁴ Os	2.1×10^{-7}
⁴¹ Ca	2.5×10^{-6}	²⁰⁷ Bi	1.6×10^{-7}
⁶⁰ Co	1.1×10^{-6}	²¹⁰ Pb	2.7×10^{-4}
⁵⁹ Ni	7.1×10^{-8}	²²⁶ Ra ^d	1.1×10^{-5}
⁶³ Ni	2.0×10^{-7}	²²⁸ Ra + d	1.7×10^{-5}
⁷⁹ Se	1.2×10^{-6}	²²⁷ Ac + d	3.3×10^{-5}
⁸⁷ Rb	1.9×10^{-6}	²²⁹ Th + d	5.1×10^{-6}
⁹⁰ Sr	2.0×10^{-4}	²³⁰ Th	2.6×10^{-7}
⁹³ Zr ^c	4.9×10^{-9}	²³² Th ^c	1.3×10^{-6}
^{93m} Nb	1.5×10^{-8}	²³¹ Pa ^f	1.6×10^{-5}
⁹⁴ Nb	2.0×10^{-7}	²³² U + d	3.5×10^{-5}
⁹⁹ Tc	1.3×10^{-5}	²³³ U	6.4×10^{-6}
¹⁰⁷ Pd	3.3×10^{-8}	²³⁴ U	6.2×10^{-6}
^{113m} Cd	1.4×10^{-4}	²³⁵ U	6.0×10^{-6}
^{121m} Sn	5.4×10^{-8}	²³⁶ U	6.0×10^{-6}
¹²⁶ Sn + d	8.8×10^{-7}	²³⁸ U	5.5×10^{-6}
¹²⁹ I	8.0×10^{-5}	²³⁷ Np	2.5×10^{-4}
¹³⁵ Cs	1.2×10^{-6}	²³⁸ Pu	7.9×10^{-7}

Table G.18 (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
²³⁹ Pu	8.6×10^{-7}	²⁴⁴ Cm	1.7×10^{-7}
²⁴⁰ Pu	8.6×10^{-7}	²⁴⁵ Cm	3.1×10^{-7}
²⁴¹ Pu	1.7×10^{-8}	²⁴⁶ Cm	3.1×10^{-7}
²⁴² Pu	8.4×10^{-7}	²⁴⁷ Cm	2.9×10^{-7}
²⁴⁴ Pu	8.2×10^{-7}	²⁴⁸ Cm	1.2×10^{-6}
²⁴¹ Am	5.1×10^{-6}	²⁴⁹ Cf	4.0×10^{-7}
^{242m} Am ^g	5.0×10^{-6}	²⁵⁰ Cf	1.8×10^{-7}
²⁴³ Am	5.1×10^{-6}	²⁵¹ Cf	4.1×10^{-7}
²⁴³ Cm	2.1×10^{-7}		

^aValues give 50-year committed effective dose equivalents from one-year intake of vegetables.

^b"n + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^ePossible contributions to dose from ²²⁸Ra decay product are not included.

^fPossible contributions to dose from ²²⁷Ac decay product are not included.

^gPossible contributions to dose from ²³⁸Pu decay product are not included.

garden is consistent with data obtained near the Savannah River Site (Hamby 1992), which should be reasonably representative of data near the ORR and the assumption that half of an intruder's entire intakes of vegetables would be obtained from the home garden. An assumption that an intruder's entire intakes of vegetables would be obtained from the home garden is unreasonably conservative.

The model for estimating dose from the vegetable pathway is summarized in Table G.18. The annual dose per unit concentration of a radionuclide in exhumed waste from a disposal unit at the time intrusion occurs is based on the model and parameter values described above and the ingestion dose conversion factors given in Table G.2.

Soil Ingestion Pathway. The annual committed effective dose equivalent (rem per year) from direct ingestion of radionuclide *i* in contaminated soil (*s*) is given by

$$H_{is} = C_{is} U_s D_i, \quad (\text{G.11})$$

where

- C_{is} = concentration of radionuclide i in soil in vegetable garden ($\mu\text{Ci/kg}$),
 U_s = annual consumption of contaminated soil (kg/year),
 D_i = dose conversion factor for ingestion of radionuclide i ($\text{rem}/\mu\text{Ci ingested}$).

In this analysis, ingestion of contaminated soil is assumed to occur as a result of incomplete washing of vegetables from the garden before consumption. At humid sites with extensive vegetation, such as the ORR, direct ingestion of contaminated soil from sources other than the garden should be relatively unimportant for an average adult. Radionuclide concentrations in soil in the vegetable garden are given by

$$C_{is} = f_s C_{it} / \rho_s, \quad (\text{G.12})$$

where

- C_{it} = concentration of radionuclide i in disposal units ($\mu\text{Ci/m}^3$),
 f_s = dilution factor for mixing of materials exhumed from disposal units into soil in vegetable garden,
 ρ_s = density of soil (kg/m^3).

In implementing the model, a dilution factor for mixing of exhumed materials from the disposal units into native soil in the vegetable garden of 0.2 and a density of soil of $1,400 \text{ kg/m}^3$ are assumed, as in the model for the vegetable pathway described above. The daily consumption of contaminated soil from the vegetable garden is assumed to be 0.1 g, which corresponds to an annual consumption of 0.037 kg. The assumed consumption of contaminated soil is appropriate for average adults (EPA 1989).

The model for estimating dose from the soil ingestion pathway is summarized in Table G.19. The annual dose per unit concentration of a radionuclide in exhumed waste from a disposal unit at the time intrusion occurs is based on the model and parameter values described above and the ingestion dose conversion factors given in Table G.2.

External Exposure Pathways. External exposure of inadvertent intruders is assumed to occur while working in the vegetable garden containing contaminated soil and while residing in the home on top of disposed waste. For external exposure (e) to contaminated soil while working in the vegetable garden, the annual effective dose equivalent (rem per year) from radionuclide i is given by

$$H_{ie} = C_{is} U_s D_{is}, \quad (\text{G.13})$$

where

- C_{is} = concentration of radionuclide i in soil in vegetable garden ($\mu\text{Ci/m}^3$),

- U_s = fraction of the year during which external exposure to contaminated soil in vegetable garden occurs,
 D_{is} = dose conversion factor for external exposure to radionuclide i in garden soil (rem/year per $\mu\text{Ci}/\text{m}^3$).

As in the models for the vegetable and soil ingestion pathways described previously, the radionuclide concentrations in soil in the vegetable garden are given by

$$C_{is} = f_s C_{it}, \quad (\text{G.14})$$

where

- C_{it} = concentration of radionuclide i in disposal units ($\mu\text{Ci}/\text{m}^3$),
 f_s = dilution factor for mixing of materials exhumed from disposal units into soil in vegetable garden.

In implementing the model, a dilution factor for mixing of materials exhumed from the disposal units into native soil in the vegetable garden of 0.2 is assumed, as in the models for the vegetable and soil ingestion pathways described above. The fraction of the year during which external exposure while working in the garden occurs is assumed to be 0.01 (Oztunali et al. 1981); i.e., the assumed exposure time is about 100 h/year.

The model for estimating external dose while working in the garden is summarized in Table G.20. The annual dose per unit concentration of a radionuclide in exhumed waste in a disposal unit at the time intrusion occurs is based on the model and parameter values described above and the external dose-rate conversion factors given in Table G.5.

For external exposure to exposed waste during indoor residence on top of disposal units, the annual effective dose equivalent (rem per year) from radionuclide i is given by

$$H_{ie} = C_{it} U_i D_{it} S, \quad (\text{G.15})$$

where

- C_{it} = concentration of radionuclide i in disposal units ($\mu\text{Ci}/\text{m}^3$),
 U_i = fraction of the year during which external exposure to waste in disposal units during indoor residence occurs,
 D_{it} = dose conversion factor for external exposure to radionuclide i in disposal units (rem/y per $\mu\text{Ci}/\text{m}^3$),
 S = shielding factor for radionuclides during indoor residence.

Table G.19. Annual effective dose equivalents to inadvertent intruders from soil ingestion pathway per unit concentration of radionuclides in disposal units for agriculture scenario^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
³ H	3.4×10^{-10}	¹³⁷ Cs	2.7×10^{-7}
¹⁰ Be	2.5×10^{-8}	¹⁵¹ Sm	2.1×10^{-9}
¹⁴ C	1.1×10^{-8}	¹⁵² Eu	3.4×10^{-8}
²⁶ Al	8.0×10^{-8}	¹⁵⁴ Eu	5.0×10^{-8}
³⁶ Cl	1.6×10^{-8}	¹⁵⁵ Eu	8.0×10^{-9}
⁴⁰ K	1.0×10^{-7}	¹⁹⁴ Os	5.8×10^{-8}
⁴¹ Ca	4.9×10^{-9}	²⁰⁷ Pb	2.9×10^{-8}
⁶⁰ Co	1.4×10^{-7}	²¹⁰ Pb + d	3.9×10^{-5}
⁵⁹ Ni	1.1×10^{-9}	²²⁶ Ra ^d	6.9×10^{-6}
⁶³ Ni	3.1×10^{-9}	²²⁸ Ra + d	1.2×10^{-5}
⁷⁹ Se	4.6×10^{-8}	²²⁷ Ac + d	7.8×10^{-5}
⁸⁷ Rb	2.6×10^{-8}	²²⁹ Th + d	2.1×10^{-5}
⁹⁰ Sr + d	8.0×10^{-7}	²³⁰ Th	2.9×10^{-6}
⁹³ Zr ^c	9.0×10^{-9}	²³² Th ^e	1.4×10^{-5}
^{93m} Nb	2.8×10^{-9}	²³¹ Pa ^f	5.8×10^{-5}
⁹⁴ Nb	3.8×10^{-8}	²³² U + d	1.1×10^{-5}
⁹⁹ Tc	8.0×10^{-9}	²³³ U	1.5×10^{-6}
¹⁰⁷ Pd	8.0×10^{-10}	²³⁴ U	1.5×10^{-6}
^{113m} Cd	8.5×10^{-7}	²³⁵ U	1.4×10^{-6}
^{121m} Sn	8.5×10^{-9}	²³⁶ U	1.4×10^{-6}
¹²⁶ Sn + d	1.1×10^{-7}	²³⁸ U + d	1.4×10^{-6}
¹²⁹ I	1.5×10^{-6}	²³⁷ Np	2.3×10^{-5}
¹³⁵ Cs	3.7×10^{-8}	²³⁸ Pu	1.7×10^{-5}

Table G.19 (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
²³⁹ Pu	1.9×10^{-5}	²⁴⁴ Cm	1.1×10^{-5}
²⁴⁰ Pu	1.9×10^{-5}	²⁴⁵ Cm	2.0×10^{-5}
²⁴¹ Pu	3.6×10^{-7}	²⁴⁶ Cm	2.0×10^{-5}
²⁴² Pu	1.8×10^{-5}	²⁴⁷ Cm	1.8×10^{-5}
²⁴⁴ Pu	1.7×10^{-5}	²⁴⁸ Cm	7.4×10^{-5}
²⁴¹ Am	1.9×10^{-5}	²⁴⁹ Cf	2.5×10^{-5}
^{242m} Am + d ^g	1.9×10^{-5}	²⁵⁰ Cf	1.1×10^{-5}
²⁴³ Am	1.9×10^{-5}	²⁵¹ Cf	2.5×10^{-5}
²⁴³ Cm	1.3×10^{-5}		

^aValues give 50-year committed effective dose equivalents from one-year intake of soil.

^b"m + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^ePossible contributions to dose from ²²⁸Ra decay product are not included.

^fPossible contributions to dose from ²²⁷Ac decay product are not included.

^gPossible contributions to dose from ²³⁸Pu decay product are not included.

The shielding factor takes into account the reduction in external dose provided by the walls and floor of the home.

In implementing the model, the fraction of the year during which exposure in the home is assumed to occur is 0.5 (Oztunali et al. 1981); i.e., the assumed exposure time is about 4000 h/year. The indoor exposure time could be as much as about a factor of 2 higher, but the assumed value is intended to be representative of an average individual residing on the disposal site. A shielding factor during indoor residence of 0.7 is assumed for all photon-emitting radionuclides (NRC 1977).

The model for estimating external dose during indoor residence is summarized in Table G.21. The annual dose per unit concentration of a radionuclide in a disposal unit at the time intrusion occurs is based on the model and parameter values described above and the external dose-rate conversion factors for "No shielding" given in Table G.6.

Table G.20. Annual effective dose equivalents to inadvertent intruders from external exposure while working in vegetable garden per unit concentration of radionuclides in disposal units for agriculture scenario

Nuclide ^a	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^a	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
²⁶ Al	1.8×10^{-5}	²³² Th ^b	—
⁴⁰ K	1.1×10^{-6}	²³¹ Pa ^c	2.2×10^{-7}
⁶⁰ Co	1.7×10^{-5}	²³² U + d	1.0×10^{-5}
⁹⁴ Nb	1.1×10^{-5}	²³⁵ U + d	9.6×10^{-7}
^{121m} Sn	2.4×10^{-9}	²³⁸ U + d	1.5×10^{-7}
¹²⁶ Sn + d	1.3×10^{-5}	²³⁷ Np + d	1.3×10^{-6}
¹²⁹ I	1.6×10^{-8}	²⁴⁴ Pu + d	2.2×10^{-6}
¹³⁷ Cs + d	3.8×10^{-6}	²⁴¹ Am	5.4×10^{-8}
¹⁵² Eu	7.6×10^{-6}	^{242m} Am + d	8.2×10^{-8}
¹⁵⁴ Eu	8.2×10^{-6}	²⁴³ Am + d	1.1×10^{-6}
¹⁵⁵ Eu	2.2×10^{-7}	²⁴³ Cm	7.0×10^{-7}
²⁰⁷ Bi	1.0×10^{-5}	²⁴⁵ Cm	4.2×10^{-7}
²²⁶ Ra + d	1.2×10^{-5}	²⁴⁷ Cm + d	2.1×10^{-6}
²²⁸ Ra + d	1.7×10^{-5}	²⁴⁹ Cf	2.2×10^{-6}
²²⁷ Ac + d	2.0×10^{-6}	²⁵¹ Cf	6.4×10^{-7}
²²⁹ Th + d	1.9×10^{-6}		

^a" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^bPossible contributions to dose from ²²⁸Ra decay product are not included.

^cPossible contributions to dose from ²²⁷Ac decay product are not included.

Table G.21. Annual effective dose equivalents to inadvertent intruders from external exposure during indoor residence per unit concentration of radionuclides in disposal units for agriculture scenario

Nuclide ^a	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^a	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
²⁶ Al	3.9×10^{-3}	²³² Th ^b	—
⁴⁰ K	2.3×10^{-4}	²³¹ Pa ^c	4.2×10^{-5}
⁶⁰ Co	3.5×10^{-3}	²³² U + d	2.2×10^{-3}
⁹⁴ Nb	2.1×10^{-3}	²³⁵ U + d	1.6×10^{-4}
^{121m} Sn	4.2×10^{-7}	²³⁸ U + d	2.9×10^{-3}
¹²⁶ Sn + d	2.6×10^{-3}	²³⁷ Np + d	2.4×10^{-4}
¹²⁹ I	2.8×10^{-6}	²⁴⁴ Pu + d	4.6×10^{-4}
¹³⁷ Cs + d	7.4×10^{-4}	²⁴¹ Am	9.5×10^{-4}
¹⁵² Eu	1.5×10^{-3}	^{242m} Am + d	1.5×10^{-3}
¹⁵⁴ Eu	1.7×10^{-3}	²⁴³ Am + d	2.0×10^{-4}
¹⁵⁵ Eu	3.9×10^{-5}	²⁴³ Cm	1.3×10^{-4}
²⁰⁷ Pb	2.1×10^{-3}	²⁴⁵ Cm	7.4×10^{-3}
²²⁶ Ra + d	2.4×10^{-3}	²⁴⁷ Cm + d	4.0×10^{-4}
²²⁸ Ra + d	3.5×10^{-3}	²⁴⁹ Cf	4.2×10^{-4}
²²⁷ Ac + d	3.7×10^{-4}	²⁵¹ Cf	1.2×10^{-4}
²²⁹ Th + d	3.5×10^{-4}		

^a" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^bPossible contributions to dose from ²²⁸Ra decay product are not included.

^cPossible contributions to dose from ²²⁷Ac decay product are not included.

Inhalation Pathways. Inhalation exposure of inadvertent intruders is assumed to occur while working in the vegetable garden containing contaminated soil and while residing in the home on top of disposed waste. For either exposure pathway, the annual committed effective dose equivalent (rem per year) from inhalation of radionuclide *i* suspended into air (*a*) in particulate form is given by

$$H_{ia} = C_{ia} f_a U_a D_i, \quad (\text{G.16})$$

where

- C_{ia} = concentration of radionuclide i in air ($\mu\text{Ci}/\text{m}^3$),
- f_a = fraction of the year during which inhalation exposure occurs,
- U_a = annual air intake (m^3/year),
- D_i = dose conversion factor for inhalation of radionuclide i ($\text{rem}/\mu\text{Ci}$ inhaled).

Concentrations of suspended radionuclides in particulate form in air are estimated using a mass-loading model (Anspaugh et al. 1975), which is based on observations of airborne concentrations of naturally occurring materials, such as uranium and thorium, relative to their concentrations in surface soil. In a mass-loading model, airborne concentrations of radionuclides are given by

$$C_{ia} = C_{is} L_a / \rho_s, \quad (\text{G.17})$$

where

- C_{is} = concentration of radionuclide i in surface soil ($\mu\text{Ci}/\text{m}^3$),
- L_a = atmospheric mass loading of surface soil (kg/m^3),
- ρ_s = density of soil (kg/m^3).

This model is applied to all radionuclides except isotopes of radon and their short-lived decay products. The model for estimating inhalation dose due to radon released from contaminated soil or disposal units is described later in this section.

For inhalation exposure while working in the vegetable garden, the concentration of radionuclide i in soil again is given by

$$C_{is} = f_s C_{iu}, \quad (\text{G.18})$$

where

- C_{iu} = concentration of radionuclide i in disposal units ($\mu\text{Ci}/\text{m}^3$),
- f_s = dilution factor for mixing of materials exhumed from disposal units into soil in vegetable garden.

In implementing the model, a dilution factor for mixing of materials exhumed from disposal units into native soil in the garden of 0.2, a density of soil of $1,400 \text{ kg}/\text{m}^3$, and a fraction of the year during which exposure while working in the garden occurs of 0.01 (i.e., about 100 h/year) again are assumed, as in the model for external exposure while working in the garden described above. The annual air intake (breathing rate) is assumed to be $8,000 \text{ m}^3/\text{year}$ (NRC 1977). Finally, as described

below, the atmospheric mass loading of contaminated soil while working in the garden is assumed to be 10^{-7} kg/m³.

The assumed atmospheric mass loading of contaminated soil while working in the vegetable garden of 10^{-7} kg/m³ is somewhat greater than the average background dust loading for non-urban areas in the United States of about 4×10^{-8} kg/m³ (Anspaugh et al. 1975). Furthermore, the assumed value is in good agreement with an average dust loading of 6×10^{-8} kg/m³ measured above two agricultural fields at the Savannah River Site (Shinn et al. 1982), where meteorological conditions and soil moisture levels should be similar to those on the ORR. The choice of an atmospheric mass loading for this exposure pathway is based primarily on these data, but it also takes into account the following considerations.

Some gardening activities (e.g., hoeing or tilling) should increase atmospheric concentrations of suspended soil well above average background levels. However, for the following reasons, the average concentration during all gardening activities probably would not be much greater than the average background level in the United States. First, the average background level of suspended soil originating from the ORR should be substantially lower than the average level in the United States, due to the high annual precipitation, extensive vegetation, and low average wind speed at the site. Second, at any location, airborne concentrations of suspended soil consist of material originating from a wide area, not just from the location where inhalation exposures occur. Finally, the model assumes that all suspended soil particles are respirable. However, particularly during more vigorous gardening activities that could result in unusually high atmospheric mass loadings, some particles are likely to be too large to be respirable.

Taking the available data and the other factors described above into account, the choice of 10^{-7} kg/m³ to represent the average atmospheric mass loading during gardening activities on the ORR appears to be a reasonable assumption for this highly uncertain parameter.

The model for estimating inhalation dose while working in the garden is summarized in Table G.22. The annual dose per unit concentration of a radionuclide in exhumed waste in a disposal unit at the time intrusion occurs is based on the model and parameter values described above and the inhalation dose conversion factors given in Table G.3. Again, the results for isotopes of radon are estimated using a model described later in this section.

For inhalation exposures while residing in the home, the airborne concentration of radionuclide i is given by

$$C_{ia} = C_{ii} L_a / \rho_s, \quad (G.19)$$

where

$$\begin{aligned} C_{ii} &= \text{concentration of radionuclide } i \text{ in disposal units } (\mu\text{Ci/m}^3), \\ L_a &= \text{mass loading of soil in the atmosphere (kg/m}^3), \\ \rho_s &= \text{density of soil (kg/m}^3). \end{aligned}$$

Table G.22. Annual effective dose equivalents to inadvertent intruders from inhalation exposure while working in vegetable garden per unit concentration of radionuclides in disposal units for agriculture scenario^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
³ H	1.1×10^{-13}	¹³⁷ Cs	3.5×10^{-11}
¹⁰ Be	3.9×10^{-10}	¹⁵¹ Sm	3.3×10^{-11}
¹⁴ C	2.5×10^{-12}	¹⁵² Eu	2.4×10^{-10}
²⁶ Al	8.8×10^{-11}	¹⁵⁴ Eu	3.2×10^{-10}
³⁶ Cl	2.4×10^{-11}	¹⁵⁵ Eu	4.5×10^{-11}
⁴⁰ K	1.3×10^{-11}	¹⁹⁴ Os	7.4×10^{-10}
⁴¹ Ca	1.4×10^{-12}	²⁰⁷ Bi	2.2×10^{-11}
⁶⁰ Co	2.4×10^{-10}	²¹⁰ Pb + d	2.5×10^{-8}
⁵⁹ Ni	1.4×10^{-12}	²²⁶ Ra ^{d,e}	9.5×10^{-9}
⁶³ Ni	3.4×10^{-12}	²²² Rn	1.3×10^{-4}
⁷⁹ Se	7.2×10^{-12}	²²⁸ Ra + d ^e	3.7×10^{-7}
⁸⁷ Rb	3.5×10^{-12}	²²⁰ Rn	2.1×10^{-5}
⁹⁰ Sr + d	2.8×10^{-10}	²²⁷ Ac	7.4×10^{-6}
⁹³ Zr ^c	3.5×10^{-10}	²²⁹ Th	1.9×10^{-6}
^{93m} Nb	3.2×10^{-11}	²³⁰ Th	2.9×10^{-7}
⁹⁴ Nb	4.5×10^{-10}	²³² Th ^f	1.3×10^{-6}
⁹⁹ Tc	1.1×10^{-12}	²³¹ Pa ^g	1.4×10^{-6}
¹⁰⁷ Pd	1.3×10^{-11}	²³² U + d ^e	1.1×10^{-6}
^{113m} Cd	1.7×10^{-9}	²²⁰ Rn	2.1×10^{-5}
^{121m} Sn	1.3×10^{-11}	²³³ U	1.5×10^{-7}
¹²⁶ Sn + d	1.1×10^{-10}	²³⁴ U	1.4×10^{-7}
¹²⁹ I	1.9×10^{-10}	²³⁵ U	1.3×10^{-7}
¹³⁵ Cs	5.1×10^{-12}	²³⁶ U	1.4×10^{-7}

Table G.22 (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
²³⁸ U	1.3×10^{-7}	²⁴³ Am	4.8×10^{-7}
²³⁷ Np	5.9×10^{-7}	²⁴³ Cm	3.4×10^{-7}
²³⁸ Pu	4.3×10^{-7}	²⁴⁴ Cm	2.8×10^{-7}
²³⁹ Pu	4.7×10^{-7}	²⁴⁵ Cm	5.1×10^{-7}
²⁴⁰ Pu	4.7×10^{-7}	²⁴⁶ Cm	5.0×10^{-7}
²⁴¹ Pu	9.1×10^{-9}	²⁴⁷ Cm	4.5×10^{-7}
²⁴² Pu	4.5×10^{-7}	²⁴⁸ Cm	1.9×10^{-6}
²⁴⁴ Pu	4.4×10^{-7}	²⁴⁹ Cf	6.4×10^{-7}
²⁴¹ Am	4.8×10^{-7}	²⁵⁰ Cf	2.9×10^{-7}
^{242m} Am + d ^h	4.9×10^{-7}	²⁵¹ Cf	6.5×10^{-7}

^aValues give 50-year committed effective dose equivalents from one-year intake of air.

^b"m" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^eContribution to dose from radon decay product is listed separately.

^fPossible contributions to dose from ²²⁸Ra decay product are not included.

^gPossible contributions to dose from ²²⁷Ac decay product are not included.

^hPossible contributions to dose from ²³⁸Pu decay product are not included.

In implementing the model, a density of soil of $1,400 \text{ kg}/\text{m}^3$, a fraction of the year during which exposure in the home occurs of 0.5 (i.e., about 4,000 h/year), and an annual air intake of $8,000 \text{ m}^3/\text{year}$ again are assumed, as in the models for other exposure pathways. The atmospheric mass loading of contaminated soil at the location of the home on top of disposal units is assumed to be $10^{-8} \text{ kg}/\text{m}^3$, which is approximately one-fourth of the average dust loading in the United States (Anspaugh et al. 1975). The assumed atmospheric mass loading is based on the following considerations.

From the previous discussion of the atmospheric mass loading of contaminated soil while working in the vegetable garden, it seems unreasonable to assume that the mass loading of largely undisturbed surface soil on the ORR during indoor residence would be as high as the average dust loading in the United States. The assumption that the average mass loading at the disposal site is one-fourth of the average background level in the United States is intended to take into account the

abundant precipitation, extensive vegetation, and low average wind speed at the disposal site, as well as the presence of uncontaminated soil suspended from other locations. In addition, the model for inhalation exposure indoors does not reflect the possibility that indoor concentrations of suspended soil particles may be somewhat less than the concentrations outdoors (e.g., due to filtering of air by doors and windows).

The model for estimating inhalation dose during indoor residence is summarized in Table G.23. For all radionuclides except isotopes of radon, the annual dose per unit concentration in disposal units at the time intrusion occurs is based on the model and parameter values described above and the inhalation dose conversion factors given in Table G.3. The results for radon are estimated using the model described below.

In this analysis, doses from inhalation of isotopes of radon and their short-lived decay products (1) while working in the vegetable garden containing contaminated soil or (2) while residing in a home on top of exposed waste in disposal units are estimated using a natural analog model. Specifically, known average doses from indoor and outdoor exposure to ^{222}Rn and ^{220}Rn , which result from known average concentrations of their respective long-lived parent radionuclides ^{226}Ra and ^{232}Th in surface soil, are used to estimate doses from the radon isotopes per unit concentration of the parent radionuclides in disposed waste for the inhalation pathways of concern for the agriculture scenario. The analysis based on the natural analog model proceeds as follows.

The National Council on Radiation Protection and Measurements (NCRP 1987) has estimated that the average effective dose equivalent from exposure to radon in the United States is about 0.2 rem per year. This estimate applies only to exposure to ^{222}Rn and its short-lived decay products in homes and, furthermore, assumes continuous residence indoors (NCRP 1987). The estimated dose from ^{222}Rn indoors results from an average concentration of the parent radionuclide ^{226}Ra in surface soil of about 0.6 pCi/g (NCRP 1984), which, for an average bulk density of soil of 1.4 g/cm^3 (Baes and Sharp 1983), corresponds to a concentration of $0.84 \text{ } \mu\text{Ci/m}^3$. Therefore, for continuous residence indoors, the annual effective dose equivalent from exposure to ^{222}Rn and its short-lived decay products per unit concentration of ^{226}Ra in surface soil is estimated as

^{222}Rn , continuous residence indoors –

$$(0.2 \text{ rem/year}) / (0.84 \text{ } \mu\text{Ci/m}^3) = 0.24 \text{ rem/year per } \mu\text{Ci/m}^3.$$

For application to the agriculture scenario, the average dose from ^{222}Rn during continuous residence indoors can be used to estimate the dose from inhalation during indoor residence in a home on top of exposed waste in disposal units containing ^{226}Ra by taking into account the fraction of the year during which residence in the home at the disposal site occurs. As described previously, this factor is assumed to be 0.5. Therefore, for inhalation exposure while residing in a home on top of exposed waste in disposal units, the annual effective dose equivalent from ^{222}Rn and its short-lived decay products per unit concentration of ^{226}Ra in the disposal units is estimated as

^{222}Rn , indoor residence in agriculture scenario –

$$(0.24 \text{ rem/year per } \mu\text{Ci/m}^3)(0.5) = 0.12 \text{ rem/year per } \mu\text{Ci/m}^3.$$

This dose estimate is given in Table G.23.

Table G.23. Annual effective dose equivalents to inadvertent intruders from inhalation exposure during indoor residence per unit concentration of radionuclides in disposal units for agriculture scenario^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci/m}^3$)
^3H	2.8×10^{-12}	^{137}Cs	9.3×10^{-10}
^{10}Be	1.0×10^{-8}	^{151}Sm	8.7×10^{-10}
^{14}C	6.1×10^{-11}	^{152}Eu	6.4×10^{-9}
^{26}Al	2.3×10^{-9}	^{154}Eu	8.4×10^{-9}
^{36}Cl	6.4×10^{-10}	^{155}Eu	1.2×10^{-9}
^{40}K	3.5×10^{-10}	^{194}Os	1.9×10^{-8}
^{41}Ca	3.8×10^{-11}	^{207}Bi	5.8×10^{-10}
^{60}Co	6.4×10^{-9}	$^{210}\text{Pb} + \text{d}$	6.7×10^{-7}
^{59}Ni	3.8×10^{-11}	$^{226}\text{Ra}^{d,e}$	2.5×10^{-7}
^{63}Ni	9.0×10^{-11}	^{222}Rn	1.2×10^{-1}
^{79}Se	1.9×10^{-10}	$^{228}\text{Ra} + \text{d}^e$	9.9×10^{-6}
^{87}Rb	9.3×10^{-11}	^{220}Rn	1.0×10^{-2}
$^{90}\text{Sr} + \text{d}$	7.3×10^{-9}	^{227}Ac	1.9×10^{-4}
$^{93}\text{Zr}^c$	9.3×10^{-9}	^{229}Th	4.9×10^{-5}
$^{93\text{m}}\text{Nb}$	8.4×10^{-10}	^{230}Th	7.5×10^{-6}
^{94}Nb	1.2×10^{-8}	$^{232}\text{Th}^f$	3.5×10^{-5}
^{99}Tc	2.9×10^{-11}	$^{231}\text{Pa}^g$	3.8×10^{-5}
^{107}Pd	3.5×10^{-10}	$^{232}\text{U} + \text{d}^e$	2.9×10^{-5}

Table G.23 (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
^{113m} Cd	4.4×10^{-8}	²²⁰ Rn	1.0×10^{-2}
^{121m} Sn	3.5×10^{-10}	²³³ U	4.1×10^{-6}
¹²⁶ Sn + d	2.9×10^{-9}	²³⁴ U	3.8×10^{-6}
¹²⁹ I	4.9×10^{-9}	²³⁵ U	3.5×10^{-6}
¹³⁵ Cs	1.3×10^{-10}	²³⁶ U	3.8×10^{-6}
²³⁸ U	3.5×10^{-6}	²⁴³ Am	1.3×10^{-5}
²³⁷ Np	1.6×10^{-5}	²⁴³ Cm	9.0×10^{-6}
²³⁸ Pu	1.1×10^{-5}	²⁴⁴ Cm	7.3×10^{-6}
²³⁹ Pu	1.2×10^{-5}	²⁴⁵ Cm	1.3×10^{-5}
²⁴⁰ Pu	1.2×10^{-5}	²⁴⁶ Cm	1.3×10^{-5}
²⁴¹ Pu	2.4×10^{-7}	²⁴⁷ Cm	1.2×10^{-5}
²⁴² Pu	1.2×10^{-5}	²⁴⁸ Cm	4.9×10^{-5}
²⁴⁴ Pu	1.2×10^{-5}	²⁴⁹ Cf	1.7×10^{-5}
²⁴¹ Am	1.3×10^{-5}	²⁵⁰ Cf	7.5×10^{-6}
^{242m} Am + d ^f	1.3×10^{-5}	²⁵¹ Cf	1.7×10^{-5}

^aValues give 50-year committed effective dose equivalents from one-year intake of air.

^b" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^eContribution to dose from radon decay product is listed separately.

^fPossible contributions to dose from ²²⁸Ra decay product are not included.

^gPossible contributions to dose from ²²⁷Ac decay product are not included.

^hPossible contributions to dose from ²³⁸Pu decay product are not included.

The agriculture scenario also considers inhalation doses while working in a vegetable garden containing contaminated soil. The inhalation dose from ²²²Rn while working in the vegetable garden contaminated with ²²⁶Ra is estimated as follows. The United Nations Scientific Committee on the Effects of Atomic Radiation (UNSCEAR 1988) has estimated that, for continuous exposure, the average effective dose equivalent from outdoor ²²²Rn would be about 28% of the average dose from

indoor ^{222}Rn . Therefore, for continuous exposure outdoors, the annual effective dose equivalent from ^{222}Rn and its short-lived decay products per unit concentration of ^{226}Ra in surface soil is estimated from the previous result for continuous indoor residence as:

^{222}Rn , continuous exposure outdoors –

$$(0.24 \text{ rem/year per } \mu\text{Ci/m}^3)(0.28) = 0.067 \text{ rem/year per } \mu\text{Ci/m}^3.$$

The dose from ^{222}Rn during continuous exposure outdoors can be used to estimate the dose from inhalation while working in the vegetable garden containing ^{226}Ra by applying two corrections. The first is the fraction of the year that an intruder spends working in the garden, which is assumed to be 0.01. The second is the dilution factor for mixing of radionuclides in exhumed waste materials from disposal units into soil in the garden, which is assumed to be 0.2. Therefore, for inhalation exposure while working in the vegetable garden, the annual effective dose equivalent from ^{222}Rn and its short-lived decay products per unit concentration of ^{226}Ra in exhumed waste is estimated as:

^{222}Rn , exposure in vegetable garden in agriculture scenario –

$$(0.067 \text{ rem/year per } \mu\text{Ci/m}^3)(0.01)(0.2) = 1.3 \times 10^{-4} \text{ rem/year per } \mu\text{Ci/m}^3.$$

This dose estimate is given in Table G.22.

The dose estimates for inhalation exposure to ^{220}Rn and its short-lived decay products during indoor residence and while working in the vegetable garden are obtained from the data on average doses from indoor and outdoor exposure to ^{222}Rn presented above and the following information. First, for continuous residence, the average dose from indoor ^{220}Rn is estimated to be about 14% of the average dose from indoor ^{222}Rn , and the average dose from outdoor ^{220}Rn is estimated to be about 26% of the average dose from outdoor ^{222}Rn (UNSCEAR 1988). Second, the estimated doses from ^{220}Rn result from an average concentration of the long-lived parent radionuclide ^{232}Th in surface soil of about 1 pCi/g (NCRP 1984), which, for an average bulk density of soil of 1.4 g/cm³ (Baes and Sharp 1983), corresponds to a concentration of 1.4 $\mu\text{Ci/m}^3$.

Using the data on doses from ^{222}Rn for continuous residence indoors and outdoors presented previously, the data on doses from ^{220}Rn relative to the doses from ^{222}Rn and the average concentration of ^{232}Th in surface soil, which also applies to ^{226}Ra , given above, and the assumptions in the agriculture scenario for the indoor residence time, the residence time while working in the vegetable garden, and the dilution factor for mixing of exhumed waste materials in garden soil, the following dose estimates for ^{220}Rn are obtained. For inhalation exposure while residing in a home on top of exposed waste in disposal units, the annual effective dose equivalent from ^{220}Rn and its short-lived decay products per unit concentration of ^{232}Th in the disposal units is estimated as:

²²⁰Rn, indoor residence in agriculture scenario –

$$(0.2 \text{ rem/year})(0.14)(0.5)/(1.4 \text{ } \mu\text{Ci/m}^3) = 0.010 \text{ rem/year per } \mu\text{Ci/m}^3.$$

This dose estimate is given in Table G.23. For inhalation exposure while working in the vegetable garden, the annual effective dose equivalent from ²²⁰Rn and its short-lived decay products per unit concentration of ²³²Th in exhumed waste is estimated as:

²²⁰Rn, exposure in vegetable garden in agriculture scenario –

$$(0.2 \text{ rem/year})(0.28)(0.26)(0.01)(0.2)/(1.4 \text{ } \mu\text{Ci/m}^3) = 2.1 \times 10^{-5} \text{ rem/year per } \mu\text{Ci/m}^3.$$

This dose estimate is given in Table G.22. The dose estimates for ²²⁰Rn during indoor residence and while working in the vegetable garden given above also apply to ²²⁰Rn produced in the decay of ²³²U.

All Exposure Pathways. For the agriculture scenario, the annual doses to an inadvertent intruder from all exposure pathways per unit concentration of radionuclides in disposal units at the time intrusion is assumed to occur are summarized in Table G.24. As described below, the total dose from all exposure pathways involving contaminated soil in the vegetable garden is presented separately from the total dose from all exposure pathways involving residence on top of exposed waste in the disposal units.

The different exposure pathways assumed in the agriculture scenario can be divided into those involving exposure to contaminated soil in the intruder's vegetable garden (i.e., vegetable, soil ingestion, external exposure, and inhalation pathways) and those involving exposure to waste during indoor residence on top of exposed waste in the disposal units (i.e., external exposure and inhalation pathways). Calculating the total dose as the sum of the doses from all exposure pathways would be appropriate only if the ratio of the volume of radioactive waste to the volume of uncontaminated waste materials that is excavated from the disposal units and mixed with native soil in the vegetable garden is the same as the ratio of the volume of radioactive waste to the volume of uncontaminated waste materials in the exposed disposal units themselves. However, most disposal units are constructed with thick covers of uncontaminated material, and the fraction of the volume of excavated material that contains radioactive waste would be substantially less than the corresponding fraction of the volume of exposed waste materials remaining in the disposal units after excavation. Therefore, for the agriculture scenario, it is appropriate to separate the dose per unit concentration for the vegetable, soil ingestion, external exposure, and inhalation pathways due to exposure to contaminated soil in the vegetable garden given in Table G.18, G.19, G.20, and G.22, respectively, from the dose for the external exposure and inhalation pathways during indoor residence on top of disposal units given in Tables G.21 and G.23, respectively.

Table G.24. Annual effective dose equivalents to inadvertent intruders from all exposure pathways per unit concentration of radionuclides in disposal units for agriculture scenario^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)		Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	
	Garden pathways ^c	Residence pathways ^d		Garden pathways ^c	Residence pathways ^d
³ H	4.0×10^{-6}	2.8×10^{-12}	¹²⁶ Sn + d	1.4×10^{-5}	2.6×10^{-3}
¹⁰ Be	6.5×10^{-8}	1.0×10^{-8}	¹²⁹ I	8.2×10^{-5}	2.8×10^{-6}
¹⁴ C	1.5×10^{-5}	6.1×10^{-11}	¹³⁵ Cs	1.2×10^{-6}	1.3×10^{-10}
²⁶ Al	1.8×10^{-5}	3.9×10^{-3}	¹³⁷ Cs + d	1.3×10^{-5}	7.4×10^{-4}
³⁶ Cl	1.2×10^{-3}	6.4×10^{-10}	¹⁵¹ Sm	1.1×10^{-8}	8.7×10^{-10}
⁴⁰ K	6.0×10^{-5}	2.3×10^{-4}	¹⁵² Eu	7.8×10^{-6}	1.5×10^{-3}
⁴¹ Ca	2.5×10^{-6}	3.8×10^{-11}	¹⁵⁴ Eu	8.5×10^{-6}	1.7×10^{-3}
⁶⁰ Co	1.8×10^{-5}	3.5×10^{-3}	¹⁵⁵ Eu	2.6×10^{-7}	3.9×10^{-5}
⁵⁹ Ni	7.2×10^{-8}	3.8×10^{-11}	¹⁹⁴ Os	2.7×10^{-7}	1.9×10^{-8}
⁶³ Ni	2.0×10^{-7}	9.0×10^{-11}	²⁰⁷ Bi	1.0×10^{-5}	2.1×10^{-3}
⁷⁹ Se	1.2×10^{-6}	1.9×10^{-10}	²¹⁰ Pb + d	3.1×10^{-4}	6.7×10^{-7}
⁸⁷ Rb	1.9×10^{-6}	9.3×10^{-11}	²²⁶ Ra + d ^{f,g}	3.0×10^{-5}	2.4×10^{-3}
⁹⁰ Sr + d	2.0×10^{-4}	7.3×10^{-9}	²²² Rn	1.3×10^{-4}	1.2×10^{-1}
⁹³ Zr ^e	1.4×10^{-8}	9.3×10^{-9}	²²⁸ Ra + d ^g	4.6×10^{-5}	3.5×10^{-3}
^{93m} Nb	1.8×10^{-8}	8.4×10^{-10}	²²⁰ Rn	2.1×10^{-5}	1.0×10^{-2}
⁹⁴ Nb	1.1×10^{-5}	2.1×10^{-3}	²²⁷ Ac + d	1.2×10^{-4}	5.6×10^{-4}
⁹⁹ Tc	1.3×10^{-5}	2.9×10^{-11}	²²⁹ Th + d	3.0×10^{-5}	4.0×10^{-4}
¹⁰⁷ Pd	3.4×10^{-8}	3.5×10^{-10}	²³⁰ Th	3.5×10^{-6}	7.5×10^{-6}
^{113m} Cd	1.4×10^{-4}	4.4×10^{-8}	²³² Th ^h	1.7×10^{-5}	3.5×10^{-5}
^{121m} Sn	6.5×10^{-8}	4.2×10^{-7}	²³¹ Pa ⁱ	7.6×10^{-5}	8.0×10^{-5}

Table G.24. (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)		Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	
	Garden pathways ^c	Residence pathways ^d		Garden pathways ^c	Residence pathways ^d
²³² U + d ^g	5.7×10^{-5}	2.2×10^{-3}	²⁴⁴ Pu + d	2.0×10^{-5}	4.7×10^{-4}
²²⁰ Rn	2.1×10^{-5}	1.0×10^{-2}	²⁴¹ Am	2.5×10^{-5}	2.3×10^{-5}
²³³ U	8.1×10^{-6}	4.1×10^{-6}	^{242m} Am + d ^j	2.5×10^{-5}	2.8×10^{-5}
²³⁴ U	7.8×10^{-6}	3.8×10^{-6}	²⁴³ Am + d	2.6×10^{-5}	2.1×10^{-4}
²³⁵ U + d	8.5×10^{-6}	1.6×10^{-4}	²⁴³ Cm	1.4×10^{-5}	1.4×10^{-4}
²³⁶ U	7.5×10^{-6}	3.8×10^{-6}	²⁴⁴ Cm	1.1×10^{-5}	7.3×10^{-6}
²³⁸ U + d	7.2×10^{-6}	3.3×10^{-5}	²⁴⁵ Cm	2.1×10^{-5}	8.7×10^{-5}
²³⁷ Np + d	2.7×10^{-4}	2.6×10^{-4}	²⁴⁶ Cm	2.1×10^{-5}	1.3×10^{-5}
²³⁸ Pu	1.8×10^{-5}	1.1×10^{-5}	²⁴⁷ Cm + d	2.1×10^{-5}	4.1×10^{-4}
²³⁹ Pu	2.0×10^{-5}	1.2×10^{-5}	²⁴⁸ Cm	7.7×10^{-5}	4.9×10^{-5}
²⁴⁰ Pu	2.0×10^{-5}	1.2×10^{-5}	²⁴⁹ Cf	2.8×10^{-5}	4.4×10^{-4}
²⁴¹ Pu	3.9×10^{-7}	2.4×10^{-7}	²⁵⁰ Cf	1.1×10^{-5}	7.5×10^{-6}
²⁴² Pu	1.9×10^{-5}	1.2×10^{-5}	²⁵¹ Cf	2.7×10^{-5}	1.4×10^{-4}

^aAssumed exposure pathways include vegetable, soil ingestion, external exposure, and inhalation exposure; effective dose equivalents per unit concentration in disposal units for these pathways are given in Tables G.18–G.23.

^b" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cDose from exposure to radionuclides in contaminated soil in vegetable garden is sum of doses from vegetable, soil ingestion, external exposure, and inhalation pathways given in Tables G.18, G.19, G.20, and G.22, respectively.

^dDose from exposure to radionuclides during residence on top of disposal units is sum of doses from external exposure and inhalation pathways given in Tables G.21 and G.23, respectively.

^ePossible contributions to dose from ^{93m}Nb decay product are not included.

^fPossible contributions to dose from ²¹⁰Pb decay product are not included.

^gContribution to dose from radon decay product is listed separately.

^hPossible contributions to dose from ²²⁸Ra decay product are not included.

ⁱPossible contributions to dose from ²²⁷Ac decay product are not included.

^jPossible contributions to dose from ²³⁸Pu decay product are not included.

G.5.2.2 Resident Scenario

In the resident scenario, an inadvertent intruder is assumed to construct a home on top of disposal units, as in the agriculture scenario described in the previous section. However, in digging the foundation for the home, an intruder is assumed to encounter an intact engineered barrier above the waste that cannot be penetrated by normal excavation procedures, and the intruder's home is assumed to be constructed on top of the intact barrier, rather than on top of exposed waste as in the agriculture scenario. An intruder then is assumed to receive external exposure while residing in the home on top of the disposal units, but ingestion exposures and inhalation exposures to radionuclides in particulate form are precluded by the presence of an intact engineered barrier. Inhalation of radon and its short-lived decay products during indoor residence also is assumed to be mitigated by the presence of an intact engineered barrier above the waste.

External exposure while residing in the home in the resident scenario is estimated using the model given in Eq. (G.15). As in the agriculture scenario described in the previous section, the fraction of the year during which external exposure in the home occurs is assumed to be 0.5, and the shielding factor for all photon-emitting radionuclides during indoor residence is assumed to be 0.7. The dose conversion factor for each photon-emitting radionuclide is the external dose-equivalent rate per unit concentration in the disposal units taking into account the shielding provided by the source region and the engineered barrier on top of the disposal units. For most of the disposal units in SWSA 6 constructed with engineered barriers, the intact barrier on top of the disposal units is assumed to provide shielding equivalent to a layer of soil 30 cm thick.

The model for estimating external dose in the resident scenario is summarized in Table G.25. The annual dose per unit concentration of a radionuclide in the disposal units at the time intrusion occurs is obtained from the assumed exposure time and shielding factor during indoor residence given above and the external dose-rate conversion factors for "30-cm shielding" given in Table G.6.

G.5.2.3 Discovery Scenario

In the discovery scenario, an inadvertent intruder is assumed to encounter an intact and impenetrable engineered barrier in disposal units while digging a foundation for a home, as in the resident scenario described in the previous section. However, the discovery scenario differs from the resident scenario in two respects. First, an intruder is assumed to receive an external exposure only for a short period of time while attempting to excavate at the disposal site, but the intruder does not construct a home at the location of the disposal units. Second, an intruder is assumed to receive external exposure while working at the side of disposal units, rather than on top. Thus, the discovery scenario is appropriate for disposal units in which the engineered barrier is thinner at the sides than at the top. As in the resident scenario, ingestion exposures and inhalation exposures to radionuclides in

Table G.25. Annual effective dose equivalents to inadvertent intruders per unit concentration of radionuclides in disposal units for resident scenario^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
²⁶ Al	3.3×10^{-4}	²³² Th ^c	—
⁴⁰ K	2.1×10^{-5}	²³¹ Pa ^d	4.6×10^{-7}
⁶⁰ Co	2.9×10^{-4}	²³² U + d	2.3×10^{-4}
⁹⁴ Nb	1.1×10^{-4}	²³⁵ U + d	5.6×10^{-7}
¹²⁶ Sn + d	1.1×10^{-4}	²³⁸ U + d	1.2×10^{-6}
¹³⁷ Cs + d	3.2×10^{-5}	²³⁷ Np + d	3.5×10^{-6}
¹⁵² Eu	9.8×10^{-5}	²⁴⁴ Pu + d	2.1×10^{-5}
¹⁵⁴ Eu	1.1×10^{-4}	^{242m} Am + d	2.5×10^{-7}
¹⁵⁵ Eu	2.0×10^{-8}	²⁴³ Am + d	1.1×10^{-6}
²⁰⁷ Pb	1.2×10^{-4}	²⁴³ Cm	7.7×10^{-7}
²²⁶ Ra + d	1.9×10^{-4}	²⁴⁵ Cm	9.8×10^{-8}
²²⁸ Ra + d	3.0×10^{-4}	²⁴⁷ Cm + d	8.4×10^{-6}
²²⁷ Ac + d	6.7×10^{-6}	²⁴⁹ Cf	8.1×10^{-6}
²²⁹ Th + d	1.1×10^{-5}	²⁵¹ Cf	3.9×10^{-7}

^aValues give effective dose equivalent per unit concentration of radionuclides in disposal units from one-year external exposure during indoor residence on top of waste covered with intact engineered barriers with a thickness of 30 cm.

^b"+ d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ²²⁸Ra decay product are not included.

^dPossible contributions to dose from ²²⁷Ac decay product are not included.

particulate form are precluded by the intact engineered barrier, and inhalation of radon and its short-lived decay products is assumed to be mitigated by the presence of an intact engineered barrier between the waste and the receptor location.

External exposure while digging next to disposal units in the discovery scenario is estimated using the model given in Eq. (G.13). The only parameters in the model are the time during which external exposure while excavating at the site occurs and the amount of shielding provided by the engineered barrier at the side of the disposal units.

The choice of an exposure time for the discovery scenario is rather subjective. The time spent excavating at the disposal site probably would be at least 10 h (i.e., about 0.1% of the time during a year), but seems unlikely to exceed 100 h (about 1% of the time during a year) before an intruder would decide to abandon digging at the site and move elsewhere. An exposure time of 100 hours during a year is assumed for this analysis, which should provide a prudently conservative estimate of dose.

For the different types of disposal units in SWSA 6 constructed with engineered barriers, the intact barrier at the sides of the disposal units is assumed to provide shielding equivalent to a layer of soil of thickness either 15 or 30 cm. However, doses for the discovery scenario need to be estimated only for disposal units with shielding at the sides equivalent to 15 cm of soil. For disposal units with thicker walls, the thickness of the engineered barrier at the top and sides would be the same, and the dose from the resident scenario, which assumes exposure for 50% of the time during the year, would always be much greater than the dose from the discovery scenario, which assumes exposure only for about 1% of the time during a year.

The model for estimating external dose in the discovery scenario is summarized in Table G.26. The annual dose per unit concentration of a radionuclide in the disposal units at the time intrusion occurs is obtained from the assumed exposure time given above and the external dose-rate conversion factors for "15-cm shielding" given in Table G.6.

G.5.2.4 Post-Drilling Scenario

In the post-drilling scenario, an inadvertent intruder is assumed to drill through a disposal unit (e.g., for the purpose of constructing a well for a domestic water supply), and the entire amount of drilling waste is assumed to be mixed with native soil in the intruder's vegetable garden. The pathways for chronic exposure assumed for this scenario then include (1) ingestion of vegetables grown in the contaminated garden soil, (2) direct ingestion of contaminated soil in conjunction with vegetable intakes, (3) external exposure to contaminated soil while working in the garden, and (4) inhalation of suspended activity while working in the garden.

The exposure pathways for the post-drilling scenario are conceptually the same as the corresponding four pathways for the agriculture scenario described in Sect. G.5.2.1. However, external and inhalation exposures during indoor residence do not occur in the post-drilling scenario, because all exhumed waste is assumed to be mixed with native soil in the vegetable garden and the intruder's home is not located on top of any disposal units. Therefore, the models given by

Table G.26. Effective dose equivalents to inadvertent intruders per unit concentration of radionuclides in disposal units for discovery scenario^a

Nuclide ^b	Dose (rem per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Dose (rem per $\mu\text{Ci}/\text{m}^3$)
²⁶ Al	3.0×10^{-5}	²³¹ Pa ^d	1.1×10^{-7}
⁴⁰ K	1.9×10^{-6}	²³² U + d	1.8×10^{-5}
⁶⁰ Co	2.9×10^{-5}	²³⁵ U + d	2.5×10^{-7}
⁹⁴ Nb	1.4×10^{-5}	²³⁸ U + d	1.4×10^{-7}
¹²⁶ Sn + d	1.5×10^{-5}	²³⁷ Np + d	8.1×10^{-7}
¹³⁷ Cs + d	4.5×10^{-6}	²⁴⁴ Pu + d	2.7×10^{-6}
¹⁵² Eu	1.1×10^{-5}	²⁴¹ Am	3.6×10^{-10}
¹⁵⁴ Eu	1.2×10^{-5}	^{242m} Am + d	3.4×10^{-8}
¹⁵⁵ Eu	2.0×10^{-8}	²⁴³ Am + d	3.4×10^{-7}
²⁰⁷ Pb	1.4×10^{-5}	²⁴³ Cm	2.5×10^{-7}
²²⁶ Ra + d	1.8×10^{-5}	²⁴⁵ Cm	5.6×10^{-8}
²²⁸ Ra + d	2.7×10^{-5}	²⁴⁷ Cm + d	1.7×10^{-8}
²²⁷ Ac + d	1.3×10^{-6}	²⁴⁹ Cf	1.7×10^{-8}
²²⁹ Th + d	1.5×10^{-6}	²⁵¹ Cf	1.6×10^{-7}
²³² Th ^c	—		

^aValues give effective dose equivalent per unit concentration of radionuclides in disposal units from external exposure for about 100 h next to waste shielded by intact engineered barriers of thickness 15 cm.

^b"+ d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ²²⁸Ra decay product are not included.

^dPossible contributions to dose from ²²⁷Ac decay product are not included.

Eqs. (G.9) and (G.10) for the vegetable pathway, Eqs. (G.11) and (G.12) for the soil ingestion pathway, Eqs. (G.13) and (G.14) for external exposure while working in the garden, and Eqs. (G.16), (G.17), and (G.18) for inhalation exposure while working in the garden, as well as the natural analog model for estimating inhalation dose from exposure to radon and its short-lived decay products while working in the garden, also apply to the post-drilling scenario.

In implementing the exposure pathway models for the post-drilling scenario, most of the parameter values are the same as the values assumed for the agriculture scenario. The one exception is the dilution factor for mixing of exhumed materials from a disposal unit with native soil in the vegetable garden, which is denoted by f_s . For all exposure pathways in the post-drilling scenario, the dose per unit concentration of a radionuclide in a disposal unit is directly proportional to this dilution factor.

With the assumption in the post-drilling scenario that all drilling waste is mixed with native soil in the vegetable garden, the dilution factor for mixing of drilling waste in garden soil is given by the ratio of the volume of waste exhumed by drilling to the volume of soil into which the waste is mixed. These two volumes are estimated as follows.

The volume of radioactive waste that would be exhumed by drilling can be estimated from the thickness of buried waste in disposal units and the diameter of a well. For the different types of disposal units in SWSA 6 for which the post-drilling scenario would be relevant, the thickness of buried waste is about 2.4 m for Tumulus I and II, about 3.6 m for the IWMF, and about 4.3 m or less for the different silos and wells. The post-drilling scenario is not relevant for the biological trenches, which are constructed without engineered barriers, because the agriculture scenario discussed previously would always result in the most restrictive limits on concentrations of radionuclides for disposal. Considering that some fraction of the thickness of waste in the various disposal units contains uncontaminated backfill material, a nominal thickness of radioactive waste of 4 m is assumed for all disposal units. This value should provide a conservative estimate of the actual thickness of radioactive waste in any units. The diameter of a well drilled through the waste then is assumed to be 0.15 m (i.e., about 6 in.). This value should not underestimate the diameter of a well that would be established for domestic use on the disposal site. From the assumed thickness of buried waste and diameter of a well, the assumed volume of waste exhumed by drilling then is about 0.07 m^3 .

The intruder's vegetable garden into which all of the exhumed drilling waste is mixed is assumed to be about 200 m^2 in area. A garden of this size should be sufficient to provide half of the entire yearly intake of vegetables by an intruder, as assumed in this analysis. The waste is assumed to be mixed to a depth of 0.15 m, which is a typical depth of a plowed layer in garden soil. Therefore, the volume of soil into which the waste exhumed from the disposal units is mixed is assumed to be about 30 m^3 .

Therefore, from the estimated volume of contaminated drilling waste of 0.07 m^3 and the assumed volume into which this waste is mixed of 30 m^3 , the dilution factor, f_s , assumed in the analysis for the post-drilling scenario is 0.002. This value is 1% of the corresponding dilution factor of 0.2 assumed in the agriculture scenario. This difference reflects the much smaller volume of

waste that would be exhumed by drilling compared with the volume of material that would be exhumed in digging a foundation for a home.

In the previous analysis of the post-drilling scenario (ORNL 1994), the dilution factor for mixing of exhumed waste with native soil in the intruder's vegetable garden was assumed to be 0.02, which is a factor of 10 greater than the value derived above. In the previous analysis, the volume of contaminated material exhumed by drilling was assumed to be 0.5 m³. However, this volume, which was based on an estimate for a disposal site on the Hanford Reservation (Kennedy et al. 1983), clearly is much too large for disposal units in SWSA 6, which are limited in the thickness of buried waste by the shallow depth of the water table. The value of the dilution factor used in this analysis more properly reflects the site-specific thickness of buried waste.

For the post-drilling scenario, the annual doses to an inadvertent intruder from all exposure pathways per unit concentration of radionuclides in disposal units at the time intrusion is assumed to occur, as obtained from the models and parameter values described above, are summarized in Table G.27. Based on the comparison of the waste dilution factors, f_s , for the post-drilling and agriculture scenarios described above, and taking into account that exposure during indoor residence does not occur in the post-drilling scenario, the total dose for each radionuclide is 1% of the sum of the doses from the vegetable, soil ingestion, external exposure, and inhalation pathways for the agriculture scenario given in Tables G.18, G.19, G.20, and G.22, respectively (i.e., 1% of the dose for the garden pathways in the agriculture scenario given in Table G.24).

G.6 SUMMARY

This appendix has presented the models and data bases used in estimating effective dose equivalents to off-site individuals resulting from exposure to radionuclides in contaminated groundwater or surface waters and to inadvertent intruders resulting from direct intrusion into waste disposal units in the SWSA 6 disposal facility. In each case, particular exposure scenarios and associated exposure pathways have been assumed. For each exposure pathway, simple models for estimating dose have been developed, and doses per unit concentration of radionuclides in water or in disposal units have been estimated on the basis of assumed parameter values for the particular pathway models. In the absence of site-specific data for most parameters in the models, the assumed parameter values were based on generic data available in the literature.

For each exposure scenario, the doses per unit concentration of a radionuclide for each exposure pathway have been added to obtain the total dose per unit concentration from all relevant pathways. The following summary tables give the total dose per unit concentration of radionuclides at the time intrusion is assumed to occur for the different exposure scenarios:

- Table G.15, exposure of off-site individuals to radionuclides in contaminated groundwater;
- Table G.16, exposure of off-site individuals to radionuclides in contaminated surface waters;

Table G.27. Annual effective dose equivalents to inadvertent intruders from all exposure pathways per unit concentration of radionuclides in disposal units for post-drilling scenario^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
³ H	4.0×10^{-8}	¹³⁷ Cs + d	1.3×10^{-7}
¹⁰ Be	6.5×10^{-10}	¹⁵¹ Sm	1.1×10^{-10}
¹⁴ C	1.5×10^{-7}	¹⁵² Eu	7.8×10^{-8}
²⁶ Al	1.8×10^{-7}	¹⁵⁴ Eu	8.5×10^{-8}
³⁶ Cl	1.2×10^{-5}	¹⁵⁵ Eu	2.6×10^{-9}
⁴⁰ K	6.0×10^{-7}	¹⁹⁴ Os	2.7×10^{-9}
⁴¹ Ca	2.5×10^{-8}	²⁰⁷ Pb	1.0×10^{-7}
⁶⁰ Co	1.8×10^{-7}	²¹⁰ Pb + d	3.1×10^{-6}
⁵⁹ Ni	7.2×10^{-10}	²²⁶ Ra + d ^{d,e}	3.0×10^{-7}
⁶³ Ni	2.0×10^{-9}	²²² Rn	1.3×10^{-6}
⁷⁹ Se	1.2×10^{-8}	²²⁸ Ra + d ^e	4.6×10^{-7}
⁸⁷ Rb	1.9×10^{-8}	²²⁰ Rn	2.1×10^{-7}
⁹⁰ Sr + d	2.0×10^{-6}	²²⁷ Ac + d	1.2×10^{-6}
⁹³ Zr ^c	1.4×10^{-10}	²²⁹ Th + d	3.0×10^{-7}
^{93m} Nb	1.8×10^{-10}	²³⁰ Th	3.5×10^{-8}
⁹⁴ Nb	1.1×10^{-7}	²³² Th ^f	1.7×10^{-7}
⁹⁹ Tc	1.3×10^{-7}	²³¹ Pa ^g	7.6×10^{-7}
¹⁰⁷ Pd	3.4×10^{-10}	²³² U + d ^e	5.7×10^{-7}
^{113m} Cd	1.4×10^{-6}	²²⁰ Rn	2.1×10^{-7}
^{121m} Sn	6.5×10^{-10}	²³³ U	8.1×10^{-8}
¹²⁶ Sn + d	1.4×10^{-7}	²³⁴ U	7.8×10^{-8}
¹²⁹ I	8.2×10^{-7}	²³⁵ U + d	8.5×10^{-8}
¹³⁵ Cs	1.2×10^{-8}	²³⁶ U	7.5×10^{-8}

Table G.27 (continued)^a

Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)	Nuclide ^b	Annual dose (rem/year per $\mu\text{Ci}/\text{m}^3$)
²³⁸ U + d	7.2×10^{-8}	²⁴³ Am + d	2.6×10^{-7}
²³⁷ Np + d	2.7×10^{-6}	²⁴³ Cm	1.4×10^{-7}
²³⁸ Pu	1.8×10^{-7}	²⁴⁴ Cm	1.1×10^{-7}
²³⁹ Pu	2.0×10^{-7}	²⁴⁵ Cm	2.1×10^{-7}
²⁴⁰ Pu	2.0×10^{-7}	²⁴⁶ Cm	2.1×10^{-7}
²⁴¹ Pu	3.9×10^{-9}	²⁴⁷ Cm + d	2.1×10^{-7}
²⁴² Pu	1.9×10^{-7}	²⁴⁸ Cm	7.7×10^{-7}
²⁴⁴ Pu + d	2.0×10^{-7}	²⁴⁹ Cf	2.8×10^{-7}
²⁴¹ Am	2.5×10^{-7}	²⁵⁰ Cf	1.1×10^{-7}
^{242m} Am + d ^h	2.5×10^{-7}	²⁵¹ Cf	2.7×10^{-7}

^aValues are 1% of annual effective dose equivalents per unit concentration in disposal units for garden pathways given in Table G.24.

^b" + d" with some entries denotes shorter-lived decay products that are assumed to be in secular equilibrium with parent radionuclide; see Table G.1 for decay products and branching fractions.

^cPossible contributions to dose from ^{93m}Nb decay product are not included.

^dPossible contributions to dose from ²¹⁰Pb decay product are not included.

^eContribution to dose from radon decay product is listed separately.

^fPossible contributions to dose from ²²⁸Ra decay product are not included.

^gPossible contributions to dose from ²²⁷Ac decay product are not included.

^hPossible contributions to dose from ²³⁸Pu decay product are not included.

- Table G.24, agriculture scenario for exposure of inadvertent intruders to radionuclides in disposal units;
- Table G.25, resident scenario for exposure of inadvertent intruders to radionuclides in disposal units;
- Table G.26, discovery scenario for exposure of inadvertent intruders to radionuclides in disposal units; and
- Table G.27, post-drilling scenario for exposure of inadvertent intruders to radionuclides in disposal units.

In Table G.16, separate results are given for releases to a small surface stream (i.e., White Oak Creek) or to a large river (i.e., the Clinch River), because off-site individuals who use water from the

small surface stream for a domestic water supply are assumed to consume fish from the large river, rather than from the small stream. For assumed users of the small surface stream, the dose from the fish pathway would be based on the calculated concentrations of radionuclides in the large river and the doses per unit concentration for this pathway given in Table G.13. In Table G.24, separate results are given for the pathways in the agriculture scenario involving exposure to contaminated soil in the vegetable garden and exposure to waste in the disposal units during indoor residence, because the ratio of the volume of radioactive waste to the volume of uncontaminated waste materials that is excavated from the disposal units and mixed with native soil in the vegetable garden generally would not be the same as the ratio of the volume of radioactive waste to the volume of uncontaminated materials in the exposed disposal units, due to the layer of uncontaminated cover material on top of most disposal units. The summary results in Tables G.15 and G.25–G.27 give the total dose from all exposure pathways for each exposure scenario.

In the dose analysis for each exposure scenario and exposure pathway, some of the model parameters depend on the particular radionuclide or chemical element, whereas other parameters are the same for all radionuclides. The radionuclide- or element-specific parameter values are given in Tables G.2–G.6, G.8, G.9, G.12, and G.17. The parameter values that are independent of radionuclide are summarized in Table G.28.

For the four scenarios involving direct intrusion into solid waste in disposal units, the radionuclide concentrations to which the doses are normalized are the concentrations in the disposal units at the time intrusion is assumed to occur, rather than the concentrations at the time of disposal. The dose analyses for these scenarios presented in this appendix do not include any assumptions about the time after disposal at which intrusion occurs. Such assumptions are applied when the results of the intruder dose analyses are combined with the results of the performance assessments for the various types of disposal units, which yield predictions of the concentrations of radionuclides remaining in disposal units as a function of time after disposal.

Table G.28. Summary of radionuclide-independent parameter values used in dose analyses for off-site individuals and inadvertent intruders

Parameter description	Symbol	Parameter value
Consumption of contaminated drinking water—		
humans ^a	U_w	730 L per year
dairy cattle ^a	Q_{wm}	60 L per day
beef cattle ^a	Q_{wf}	50 L per day
Consumption of contaminated fish ^b	U_q	7 kg per year
Consumption of contaminated vegetables ^c	U_v	90 kg per year (fresh weight)
Density of water ^b	ρ_w	1 kg/L
Density of soil ^c	ρ_s	1,400 kg/m ³
Dilution factor for mixing of materials exhumed		0.2 ^d
from disposal units with native soil in	f_s	0.002 ^e
vegetable garden		
Consumption of contaminated soil ^f	U_s	0.037 kg per year
Exposure times—		
swimming ^b	U_w	1% per year
working in garden ^c	U_s, f_a	1% per year
residing in home ^f	U_r, f_a	50% per year
excavating at disposal site ^g	U_s	100 h
Shielding factor for external exposure during		
indoor residence ^f	S	0.7
Air intake (breathing rate) ^f	U_a	8,000 m ³ /year
Atmospheric mass loading of contaminated		
surface soil—		
working in garden ^c	L_a	10 ⁻⁷ kg/m ³
residing in home ^d		10 ⁻⁸ kg/m ³

^aParameter applies to exposure of off-site individuals from use of contaminated groundwater or surface waters.

^bParameter applies to exposure of off-site individuals from use of contaminated surface waters.

^cParameter applies to agriculture and post-drilling scenarios for inadvertent intruders.

^dParameter applies to agriculture scenario for inadvertent intruders.

^eParameter applies to post-drilling scenario for inadvertent intruders.

^fParameter applies to agriculture and resident scenarios for inadvertent intruders.

^gParameter applies to discovery scenario for inadvertent intruders.

G.7 REFERENCES

- Ansbaugh, L. R., J. H. Shinn, P. L. Phelps, and N. C. Kennedy. 1975. "Resuspension and Redistribution of Plutonium in Soils," *Health Physics* **29**, 571.
- Baes, C. F. III, and R. D. Sharp. 1983. "A Proposal for Estimation of Soil Leaching Constants for Use in Assessment Models," *Journal of Environmental Quality* **12**, 17.
- Baes, C. F. III, R. D. Sharp, A. L. Sjoreen, and R. W. Shor. 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture*, ORNL-5786, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Chapman, W. H., H. L. Fisher, and M. W. Pratt. 1968. *Concentration Factors of Chemical Elements in Edible Aquatic Organisms*, UCRL-50564, Lawrence Radiation Laboratory, Livermore, Calif.
- DOE (U.S. Department of Energy). 1988a. "Management of Low-Level Waste," Chap. III in *Radioactive Waste Management*, Order 5820.2A, Washington, D.C.
- DOE (U.S. Department of Energy). 1988b. *External Dose-Rate Conversion Factors for Calculation of Dose to the Public*, DOE/EH-0070, Washington, D.C.
- DOE (U.S. Department of Energy). 1996. *Interim Format and Content Guide and Standard Review Plan for U.S. Department of Energy Low-Level Waste Disposal Facility Performance Assessments* (October 31).
- Eckerman, K. F., A. B. Wolbarst, and A. C. B. Richardson. 1988. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion*, Federal Guidance Report No. 11, EPA-520/1-88-020, Oak Ridge National Laboratory and U.S. Environmental Protection Agency, Washington, D.C.
- Eckerman, K. F., and J. C. Ryman. 1993. *External Exposure to Radionuclides in Air, Water, and Soil*, Federal Guidance Report No. 12, EPA 402-R-93-081, Oak Ridge National Laboratory and U.S. Environmental Protection Agency, Washington, D.C.
- EPA (U.S. Environmental Protection Agency). 1989. *Risk Assessment Guidance for Superfund. Volume I. Human Health Evaluation Manual (Part A)*, EPA/540/1-89/002, Washington, D.C.
- Garten, C. T., Jr., E. A. Boudietti, J. R. Trabalka, R. L. Walker, and T. G. Scott. 1987. "Field Studies on the Terrestrial Behavior of Actinide Elements in East Tennessee," p. 109 in *Environmental Research on Actinide Elements*, ed. by J. E. Pinder III, J. J. Alberts, K. W. McLeod, and R. G. Schreckhise, CONF-841142, U.S. Department of Energy, Washington, D.C.
- Hamby, D. M. 1992. "Site-Specific Parameter Values for the Nuclear Regulatory Commission's Food Pathway Dose Model," *Health Physics* **62**, 136.

- ICRP (International Commission on Radiological Protection). 1979. *Limits for Intakes of Radionuclides by Workers*, ICRP Publication 30, Part 1, *Annals of the International Commission on Radiological Protection* 2, No. 3/4.
- Kennedy, W. E., Jr., B. A. Napier, and J. K. Soldat. 1983. "Advanced Disposal Systems for Transuranic Waste: Preliminary Disposal Criteria for Plutonium-239 at Hanford," *Nuclear and Chemical Waste Management* 4, 103.
- Kennedy, W. E., Jr., and D. L. Strenge. 1992. *Residual Radioactive Contamination from Decommissioning*, NUREG/CR-5512, PNL-7994, Vol. 1, Pacific Northwest Laboratory, Richland, Wash.
- Kocher, D. C. 1981. *Radioactive Decay Data Tables*, DOE/TIC-11026, U.S. Department of Energy, Washington, D.C.
- Kocher, D. C., and A. L. Sjoreen. 1985. "Dose-Rate Conversion Factors for Photon Emitters in Soil," *Health Physics* 48, 193.
- Napier, B. A., R. A. Peloquin, W. E. Kennedy, Jr., and S. M. Neuder. 1984. *Intruder Dose Pathway Analysis for the Onsite Disposal of Radioactive Waste: The ONSITE/MAXII Computer Program*, NUREG/CR-3620, PNL-4054, Pacific Northwest Laboratory, Richland, Wash.
- NCRP (National Council on Radiation Protection and Measurements). 1984. *Exposures from the Uranium Series with Emphasis on Radon and Its Daughters*, NCRP Report No. 77, NCRP, Bethesda, Md.
- NCRP (National Council on Radiation Protection and Measurements). 1987. *Exposure of the Population in the United States and Canada from Natural Background Radiation*, NCRP Report No. 94, NCRP, Bethesda, Md.
- Ng, Y. C. 1982. "A Review of Transfer Factors for Assessing the Dose from Radionuclides in Agricultural Products," *Nuclear Safety* 23, 57.
- Ng, Y. C., C. S. Colsher, D. J. Quinn, and S. E. Thompson. 1977. *Transfer Coefficients for the Prediction of the Dose to Man via the Forage-Cow-Milk Pathway from Radionuclides Released to the Biosphere*, UCRL-51939, Lawrence Livermore Laboratory, Livermore, Calif.
- NRC (U.S. Nuclear Regulatory Commission). 1977. *Regulatory Guide 1.109. Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I*, Washington, D.C.
- NRC (U.S. Nuclear Regulatory Commission). 1982. *Final Environmental Impact Statement on 10 CFR Part 61 "Licensing Requirements for Land Disposal of Radioactive Waste"*, NUREG-0945, Vol. 1, Washington, D.C.
- ORNL (Oak Ridge National Laboratory). 1994. *Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6*, ORNL-6783, Oak Ridge, Tenn.

- Oztunali, O. I., G. C. Re, P. M. Moskowitz, E. D. Picazo, and C. J. Pitt. 1981. *Data Base for Radioactive Waste Management*, NUREG/CR-1759, Vol. 3, Dames and Moore, Inc., U.S. Nuclear Regulatory Commission, Washington, D.C.
- Peterson, H. T., Jr. 1983. "Terrestrial and Aquatic Food Chain Pathways," Chap. 5 in *Radiological Assessment*, ed. J. E. Till and H. R. Meyer, NUREG/CR-3332, ORNL-5968, U.S. Nuclear Regulatory Commission and Oak Ridge National Laboratory, Washington, D. C.
- Sheppard, M. I., S. C. Sheppard, and B. D. Amiro. 1991. "Mobility and Plant Uptake of Inorganic ^{14}C and ^{14}C -Labelled PCB in Soils of High and Low Retention," *Health Physics* **61**, 481.
- Shinn, J. H., D. N. Norman, and D. D. Gay. 1982. "Plutonium Aerosol Fluxes and Pulmonary Exposure Rates During Resuspension from Bare Soils Near a Chemical Separation Facility," p. 1131 in *Precipitation Scavenging, Dry Deposition, and Resuspension*, Vol. 2, ed. by H. R. Pruppacher, R. G. Semonin, and W. G. N. Slinn, Elsevier Science Publishing Co., N. Y.
- Tuli, J. K. 1995. *Nuclear Wallet Cards*, fifth edition, Brookhaven National Laboratory, Upton, N. Y.
- UNSCEAR (United Nations Scientific Committee on the Effects of Atomic Radiation). 1988. *Sources, Effects and Risks of Ionizing Radiation*, United Nations, New York.

Appendix H

ANALYSIS OF ATMOSPHERIC EMISSIONS FROM THE SWSA 6 DISPOSAL UNITS

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Appendix H

ANALYSIS OF ATMOSPHERIC EMISSIONS FROM THE SWSA 6 DISPOSAL UNITS

The volatile radionuclides ^3H and ^{14}C may be transported from the waste disposal unit to the soil surface through a variety of mechanisms, including diffusion, burrowing animals, plant root uptake, and desiccation cracks. When these radionuclides reach the soil surface, they may become airborne and then be transported via atmospheric dispersion to receptors located downwind of the waste disposal unit.

The diffusion mechanism was selected to estimate impacts of volatile radionuclide emissions from Solid Waste Storage Area 6 (SWSA 6). The assumptions that were incorporated into this analysis to provide an upper-bound estimate of emissions that may occur due to diffusion are as follows:

- The waste is assumed to be completely mixed into the soil. Therefore, no credit is taken for reduced diffusion that would result from the various engineered barriers in the SWSA 6 disposal units. This assumption particularly bounds ^3H because of its relatively short half-life (12.3 years). The amount of ^3H in the waste at time of closure would be reduced significantly by radioactive decay during the time in which the engineered barrier maintained its integrity.
- The radionuclide is assumed to diffuse upward only. Radionuclides would actually diffuse in all directions from the waste. Therefore, approximating the flux as one-dimensional and toward the surface leads to the maximum amount that would be diffused upward.
- Diffusive flux is approximated as first-order, linear, and steady-state. Since the radionuclide concentration is the primary mechanism of diffusion, a high concentration in the waste would lead to a higher diffusive flux at the source. Near the surface, the radionuclide concentration would approach zero, and therefore flux would be significantly reduced.
- Loss of radionuclides via radioactive decay is ignored for this analysis. This again has the greatest bounding effect for ^3H due to its relatively short half-life. Only a small amount of ^{14}C (half-life of 5730 years) would decay during the active institutional control period (100 years after disposal); however, a significant amount of ^3H would decay before the end of institutional control.

These assumptions are likely to conservatively bound any combination of subsurface pathway mechanism such as burrowing animals, desiccation cracks, or plant root uptake.

Both ^3H and ^{14}C are assumed to be transported to the soil surface by diffusion in the vapor phase. Assuming that the volatile radionuclide in a disposal unit is completely bound in the pore water, the concentration of ^3H or ^{14}C in the pore water, C_{pw} (Ci/m^3), is calculated as

$$C_{pw} = \frac{C_w}{n s}, \quad (\text{H.1})$$

where

C_w = total projected activity concentration in the waste disposal unit,

n = void space of the surficial soil at atmospheric pressure, approximately equal to 0.39 (see Table D.2),

s = fraction saturation in the soil void space, assumed to be equal to 1.0 (i.e., completely saturated).

Table H.1 (^3H) and Table H.2 (^{14}C) list values of C_w for each disposal unit in SWSA 6. For all disposal units except the Interim Waste Management Facility (IWMF), values of C_w were obtained directly from Table 4.7. For the IWMF, C_w values ($9.12 \times 10^3 \text{ Ci}/\text{m}^3$ for ^3H and $1.28 \times 10^0 \text{ Ci}/\text{m}^3$ for ^{14}C) were back-calculated using the method described in this appendix in order to ensure that the performance objective for atmospheric releases (10 mrem per year from all potential exposure pathways associated with the atmospheric releases) was not exceeded at any atmospheric pathway receptor location (see Fig. 4.18). Therefore, the C_w values back-calculated for the IWMF represent the inventory limit imposed by the atmospheric pathway.

To calculate the concentration of ^3H vapor in the soil, the ratio between the concentration of ^3H in the water vapor to the ^3H concentration in the liquid water bound in the waste was determined. If the air is saturated with water at 10°C (the assumed annual average soil temperature), then the concentration of water vapor in air, or absolute humidity, is 9.2 g of water vapor per cubic meter of dry air (MMES et al. 1994). Assuming that the density of liquid water is $1.0 \times 10^6 \text{ g}/\text{m}^3$, the ratio (r) of the density of water in the air to that in the liquid phase is

$$r = \frac{9.2 \times 10^{-6} \text{ g}/\text{m}^3_{\text{(water vapor)}}}{1 \text{ g}/\text{m}^3_{\text{(liquid water)}}} \quad (\text{H.2})$$

Table H.1. SWSA 6 activity concentration and diffusive flux data used in the atmospheric pathway analysis for ^3H

Disposal unit	Waste activity concentration, C_w (Ci/m ³)	Pore water activity concentration, C_{pw} (Ci/m ³)	Vapor activity concentration, C_v (Ci/m ³)	Diffusive flux, J (Ci/m ² -s)
IWMF	$9.12 \times 10^3^a$	$2.34 \times 10^4^a$	$2.15 \times 10^{-1}^a$	$1.86 \times 10^{-8}^a$
Tumulus I	3.63×10^{-3}	9.31×10^{-3}	8.56×10^{-8}	1.84×10^{-14}
Tumulus II	2.54×10^{-3}	6.51×10^{-3}	5.99×10^{-8}	1.28×10^{-14}
Low-range silos, north (zone 8)	6.08×10^{-2}	1.56×10^{-1}	1.43×10^{-6}	3.07×10^{-13}
Low-range silos, south (zone 30)	8.81×10^{-3}	2.26×10^{-2}	2.08×10^{-7}	4.46×10^{-14}
High-range silos	2.33×10^{-2}	5.97×10^{-2}	5.50×10^{-7}	1.18×10^{-13}

^a Concentration and diffusive flux values for the IWMF were back-calculated to ensure that the performance objective for atmospheric releases (10 mrem per year from all potential exposure pathways associated with the atmospheric releases) was not exceeded at any atmospheric pathway receptor location.

Table H.2. SWSA 6 activity concentration and diffusive flux data used in the atmospheric pathway analysis for ^{14}C

Disposal unit	Waste activity concentration, C_w (Ci/m ³)	Pore water activity concentration, C_{pw} (Ci/m ³)	Vapor activity concentration, C_v (Ci/m ³)	Diffusive flux, J (Ci/m ² -s)
IWMF	1.28×10^{0a}	3.28×10^{0a}	$1.39 \times 10^{-3}^a$	$7.05 \times 10^{-11}^a$
Tumulus I	5.15×10^{-5}	1.32×10^{-4}	5.61×10^{-8}	7.05×10^{-15}
Tumulus II	7.12×10^{-6}	1.83×10^{-5}	7.76×10^{-9}	9.74×10^{-16}
Low-range silos, north (zone 8)	2.23×10^{-5}	5.72×10^{-5}	2.43×10^{-8}	3.05×10^{-15}
Low-range silos, south (zone 30)	6.32×10^{-5}	1.62×10^{-4}	6.89×10^{-8}	8.65×10^{-15}
High-range silos	9.58×10^{-5}	2.46×10^{-4}	1.04×10^{-7}	1.31×10^{-14}
Asbestos silos	1.14×10^{-5}	2.92×10^{-5}	1.24×10^{-8}	1.56×10^{-15}

^aConcentration and diffusive flux values for the IWMF were back-calculated to ensure that the performance objective for atmospheric releases (10 mrem per year from all potential exposure pathways associated with the atmospheric releases) was not exceeded at any atmospheric pathway receptor location.

The relation between the concentration of ^{14}C in the air to the ^{14}C concentration in the liquid water bound in the waste was calculated using Henry's law. This law is a linear, first-order relationship derived by approximating equilibrium conditions between dissolved gas with a particular concentration in liquid water and the same gas with a particular concentration in the air adjacent to the liquid water. All of the ^{14}C in the disposal unit is assumed to be in the volatile form of $^{14}\text{CO}_2$. It is likely that $^{14}\text{CO}_2$ would be a small fraction of the total ^{14}C , most of which would probably exist in a nonvolatile form.

For use in Henry's law, the unit mole fraction (χ) of $^{14}\text{CO}_2$ in the liquid water was determined as

$$\chi = \frac{n_{\text{CO}_2} \text{ per g of CO}_2}{n_{\text{H}_2\text{O}} \text{ per m}^3 \text{ of H}_2\text{O}} \quad (\text{H.3})$$

where

$$\begin{aligned} n_{\text{CO}_2} &= \text{number of moles in 1 g of CO}_2, \\ n_{\text{H}_2\text{O}} &= \text{number of moles in 1 m}^3 \text{ of liquid water.} \end{aligned}$$

Approximating the density of liquid water as $1.0 \times 10^6 \text{ g/m}^3$, χ is equal to 4.1×10^{-7} . The partial pressure of CO_2 in the air [p (atm)] is calculated using Henry's law:

$$p = k\chi \quad (\text{H.4})$$

where k is the Henry's law constant for CO_2 . At 10°C , k is equal to 1040 atm/mol fraction (Copper and Alley 1986).

Substituting the values for the unit mole fraction and Henry's law constant for CO_2 at 10°C , p is calculated to be $4.3 \times 10^{-4} \text{ atm}$. If the air is at atmospheric pressure, the concentration of CO_2 in the air is approximately equal to $4.3 \times 10^{-4} \text{ g/m}^3$, and therefore the ratio of CO_2 in the air to that dissolved in the water is

$$r = \frac{4.3 \times 10^{-4} \text{ g/m}^3_{(\text{CO}_2 \text{ in air})}}{1 \text{ g/m}^3_{(\text{CO}_2 \text{ dissolved in water})}} \quad (\text{H.5})$$

Because the ^3H is assumed to be completely bound with water and ^{14}C is assumed to be in the form of $^{14}\text{CO}_2$, the r values can be used to calculate the vapor concentration [C_{vap} (Ci/m^3)] of ^3H and ^{14}C as

$$C_{vap} = C_{pw} r \quad . \quad (H.6)$$

Values of C_v for each disposal unit are shown in Table H.1 and Table H.2.

The vapor flux [J (Ci/m² · s)] was approximated to be a first-order, linear process; and flux is assumed to occur from the top of the waste disposal unit (for all units except the IWMF, where flux is assumed to occur from the center of the unit) to the soil surface. Based on these approximations, the vapor flux is calculated as

$$J = \frac{D \epsilon^{4/3} C_{vap}}{x} \quad , \quad (H.7)$$

where

D = ³H diffusion coefficient in air, reported as 2.39×10^{-5} m²/s, or the ¹⁴CO₂ diffusion coefficient in air, reported as 1.40×10^{-5} m²/s (Weast and Astle 1981, as cited in MMES et al. 1994);

ϵ = soil air content, assumed to be equal to 0.046 (see Table D.2);

x = soil thickness above the waste disposal unit (m), approximately equal to 1.8 m for all units except the IWMF, and equal to 4.5 m for the IWMF.

For all units except the IWMF, the assumption that flux would occur as if the entire concentration of waste were at the top of the unit is an upper-bound approximation. Much of the waste would actually be well below the top of the disposal unit, and diffusion would occur over a greater length. Therefore, the concentration gradient and subsequent diffusive flux would be less. For the IWMF a nominal diffusion length has been selected (i.e., from the vertical center of the unit to the soil surface). Values of J calculated for each disposal unit are also shown in Table H.1 and Table H.2. These values were input as emissions for air dispersion model calculations.

The Industrial Source Complex—Version 3 Long-Term Air Dispersion Model (ISCLT3) (EPA 1995) was used to determine ambient ³H and ¹⁴C concentrations for receptors located downwind of SWSA 6. ISCLT3 is a Gaussian dispersion model approved by the U.S. Environmental Protection Agency (EPA) for refined analyses in which annual average atmospheric concentrations in simple terrain are to be calculated (40 CFR 51, App. W). The following basic assumptions are incorporated into Gaussian models (Turner 1994).

- Pollutant emissions are continuous and do not vary over time.
- The emitted mass of pollutant is conserved from the source to the receptor. Therefore, none of the pollutant is removed from the atmosphere via chemical reaction, gravitational settling, or turbulent impaction.

- Meteorological conditions are assumed to be steady state during the time in which the pollutant is transported from the source to the receptor.
- Concentration profiles in the crosswind direction (perpendicular to the path of transport) and the vertical direction are assumed to be well represented by a Gaussian distribution.

For receptors located within a few hundred meters of the source, the accuracy of Gaussian model estimates of ground-level plume centerline concentrations should be within a factor of 3 (Turner 1994).

For this analysis, each waste disposal unit was treated as an area source. Atmospheric pathway receptor locations are shown in Fig. 4.16. On-site meteorological data recorded between 1989 and 1993 were used in the analysis. Hourly data were compiled into stability array (STAR) format for use in ISCLT3.

The maximum annual average ^3H concentration calculated using ISCLT3 was about 1.2×10^4 pCi/m³, and for ^{14}C , was about 4.5×10^1 pCi/m³. Maximum annual average concentrations occurred at a receptor located to the southwest of the IWMF (see Fig. 4.18).

The performance objective for atmospheric releases, 10 mrem per year, includes doses from all potential exposure pathways associated with the atmospheric releases. The annual doses for the two volatiles of interest (^3H and ^{14}C) are calculated from four exposure pathways:

1. inhalation of airborne radionuclides,
2. ingestion of vegetation exposed to airborne radionuclides (i.e., airborne-contaminated vegetation),
3. ingestion of beef from cattle that have consumed airborne-contaminated vegetation, and
4. ingestion of milk from cattle that have consumed airborne-contaminated vegetation.

No external doses are expected, since these beta-emitting radionuclides have external dose conversion factors equal to zero (DOE 1988). The total dose from the atmospheric transport pathway is the sum of the doses from each of the four exposure pathways listed above. The equations used to calculate each exposure pathway dose are described below, and the parameter values used in these equations are listed in Table H.3. This methodology is based on a conservative specific-activity model presented in NRC Regulatory Guide 1.109 (NRC 1977). Because this is an upper-bound screening assessment, radioactive decay is not included in the calculations.

The dose from direct inhalation of volatiles is calculated using the following equation:

$$H_{inh,i} = C_{a,i} \times IR_a \times DCF_{inh,i} \quad , \quad (\text{H.8})$$

where

$$H_{inh,i} = \text{annual dose from inhalation of radionuclide } I \text{ in air (rem per year),}$$

- $C_{a,i}$ = concentration of radionuclide I in air ($\mu\text{Ci}/\text{m}^3$),
 IR_a = intake rate of air (adult inhalation rate) (m^3/year),
 $DCF_{inh,i}$ = internal dose conversion factor from inhalation for radionuclide I ($\text{rem}/\mu\text{Ci}$).

Table H.3. Parameter values used to determine dose from all potential exposure pathways associated with atmospheric emissions

Parameter	Value	Nuclide-specific value		Source
		^3H	^{14}C	
IR_a (adult average), m^3/yr	8000			ICRP 1975
$DCF_{inh,i}$, $\text{rem}/\mu\text{Ci}$		6.4×10^{-5}	$2.4 \times 10^{-5}{}^b$	EPA 1988
AH , kg/m^3	0.0092			MMES et al. 1994
$R_{ww/wa}$ (dimensionless)	0.5			NRC 1977
f_{vv} (dimensionless)	0.75			NRC 1977
$C_{air,C}$, kg/m^3	0.00016			NRC 1977; Napier et al. 1988
f_{Cv} (dimensionless)	0.11			NRC 1977
IR_v , kg/yr	90^a			MMES et al. 1994
$DCF_{ing,i}$, $\text{rem}/\mu\text{Ci}$		6.4×10^{-5}	2.1×10^{-3}	EPA 1988
$F_{h,i}$, d/kg		3.3×10^{-5}	8.5×10^{-5}	NRC 1977
$IR_{v(cow)}$, kg/yr	5694			EPA 1992
$F_{m,i}$, d/L		2.7×10^{-5}	3.3×10^{-5}	NRC 1977
IR_b , kg/yr	85			EPA 1992
IR_m , L/yr	112			EPA 1992

^a Assumes approximately half of an individual's vegetable intake is from locally grown, contaminated vegetation.

^b Carbon as CO_2 .

To calculate the ^3H doses from contaminated food, it is assumed that the concentration of ^3H in vegetation is from the air surrounding the vegetation. The concentration in the vegetation is based on the amount of water in the vegetation and the amount of ^3H that would be in the plant water. The equation for calculating the concentration of ^3H in vegetation is as follows:

$$C_{v,^3\text{H}} = \frac{C_{a,^3\text{H}}}{AH} \times R_{ww/wa} \times f_{vv} \quad (\text{H.9})$$

where

$C_{v,^3\text{H}}$ = concentration of ^3H in vegetation ($\mu\text{Ci}/\text{kg}$),

- $C_{a,^3H}$ = concentration of 3H in air ($\mu Ci/m^3$),
 AH = absolute humidity of the atmosphere (kg/m^3),
 $R_{wv/wa}$ = concentration ratio of 3H in vegetation water to 3H in atmospheric water (dimensionless),
 f_{wv} = fraction of vegetation that is water (dimensionless).

The concentration of 3H in air ($C_{a,^3H}$) divided by the absolute humidity (AH) is equivalent to the concentration in water. This, when multiplied by the fraction of water in the vegetation (f_{wv}) and the concentration ratio of 3H in vegetation water to atmospheric water ($R_{wv/wa}$), equals the concentration of 3H in the vegetation.

To calculate the doses from ^{14}C -contaminated food, the concentration of ^{14}C in vegetation is assumed to be from the air surrounding the vegetation. Also, the ratio of ^{14}C to the natural carbon in vegetation is assumed to be the same as the ratio of ^{14}C to natural carbon in the atmosphere surrounding the vegetation. For airborne releases, plants are assumed to obtain all their carbon from airborne CO_2 and animals are assumed to obtain all their carbon through ingestion of plants (NRC 1977 p. 1.109-26; Napier et al. 1988, p. 4.86).

The equation for calculating the concentration of ^{14}C in vegetation from contaminated air is

$$C_{v,^{14}C} = \frac{C_{air,^{14}C}}{C_{air,C}} \times f_{Cv} \quad (H.10)$$

where

- $C_{v,^{14}C}$ = concentration of ^{14}C in vegetation ($\mu Ci/kg$),
 $C_{air,^{14}C}$ = concentration of ^{14}C in air ($\mu Ci/m^3$),
 $C_{air,C}$ = concentration of natural carbon in air (kg/m^3),
 f_{Cv} = fraction of natural carbon in vegetation (dimensionless).

The concentration in vegetation is then used to calculate the dose from ingestion of contaminated vegetation:

$$H_{v,i} = C_{v,i} \times IR_v \times DCF_{ing,i} \quad (H.11)$$

where

- $H_{v,i}$ = annual dose from radionuclide I in vegetation (rem per year),
 $C_{v,i}$ = concentration of radionuclide I in vegetation ($\mu Ci/kg$),
 IR_v = adult intake rate of vegetables ($kg/year$),
 $DCF_{ing,i}$ = internal dose conversion factor for ingestion of radionuclide I (rem/ μCi).

As shown in Table H.3, approximately 50% of the exposed person's vegetable intake is assumed to come from the contaminated vegetation (MMES et al. 1994).

The concentration in vegetation is also used to determine the concentration in beef and milk that will be consumed by humans. The cow's vegetation consumption is assumed to be totally from grazing on fresh pasture grass contaminated with airborne radionuclides. The concentrations in beef and cow milk are calculated as follows:

$$C_{b,i} = C_{v,i} \times F_{b,i} \times IR_{v(cow)} \quad , \quad (\text{H.12})$$

and

$$C_{m,i} = C_{v,i} \times F_{m,i} \times IR_{v(cow)} \quad , \quad (\text{H.13})$$

where

- $C_{(b,m),I}$ = concentration of radionuclide I in beef (b) or dairy (m) cattle that consumed contaminated vegetation ($\mu\text{Ci/kg}$ for beef and $\mu\text{Ci/L}$ for dairy),
- $C_{v,i}$ = concentration of radionuclide I in vegetation (v) consumed by beef or dairy cattle ($\mu\text{Ci/kg}$ for beef and $\mu\text{Ci/L}$ for dairy),
- $F_{(b,m),I}$ = transfer coefficient for beef (b) (d/kg) and milk (m) (d/L) for radionuclide element I ,
- $IR_{v(cow)}$ = consumption rate of vegetation by beef or dairy cattle (kg/year).

The annual radiation doses from the ingestion of beef and milk are calculated by multiplying the concentration in each medium by the human intake rate and the radionuclide-specific internal dose conversion factor:

$$H_{b,i} = C_{b,i} \times IR_b \times DCF_{ing,i} \quad , \quad (\text{H.14})$$

and

$$H_{m,i} = C_{m,i} \times IR_m \times DCF_{ing,i} \quad , \quad (\text{H.15})$$

where

- $H_{(b,m),I}$ = annual dose from radionuclide I in beef (b) or milk (m) (rem per year),
- $C_{(b,m),I}$ = concentration of radionuclide I in beef (b) ($\mu\text{Ci/kg}$) or milk (m) ($\mu\text{Ci/L}$),
- $IR_{(b,m)}$ = adult intake rate of beef (b) (kg/year) or milk (m) (kg/L),
- $DCF_{ing,i}$ = internal dose conversion factor for ingestion of radionuclide I (rem/ μCi).

The annual dose from atmospheric releases of radionuclide I ($H_{T,i}$) is the sum of the annual doses from the four exposure pathways described above (i.e., inhalation of contaminated air and ingestion of contaminated vegetables, beef, and milk):

$$H_{T,i} = H_{inh,i} + H_{v,i} + H_{b,i} + H_{m,i} \quad (\text{H.16})$$

The annual effective dose equivalents (EDEs) from the atmospheric transport pathway, using C_a values of $1.2 \times 10^{-2} \mu\text{Ci}/\text{m}^3$ ($1.2 \times 10^4 \text{ pCi}/\text{m}^3$) and $4.5 \times 10^{-5} \mu\text{Ci}/\text{m}^3$ ($4.5 \times 10^1 \text{ pCi}/\text{m}^3$) for ^3H and ^{14}C , respectively, are presented in Table H.4. The annual doses from all exposure pathways resulting from atmospheric releases (H_T) are 10 mrem per year for both ^3H and ^{14}C . These annual doses equal the performance objective.

Table H.4. Calculated annual dose per unit concentration values and annual effective dose equivalents (EDEs) from the atmospheric transport pathway^a

	^3H		^{14}C	
	Annual EDE per unit concentration (mrem/year per $\mu\text{Ci}/\text{m}^3$)	Annual EDE ^a (mrem per year)	Annual EDE per unit concentration (mrem/year per $\mu\text{Ci}/\text{m}^3$)	Annual EDE ^a (mrem per year)
H_{inh}	512	6.14	188	0.000854
H_v	235	2.82	130,000	5.91
H_b	41.7	0.500	59,400	2.70
H_m	44.9	0.538	30,400	1.38
H_T	833	10.0	220,000	10.0

^a The calculations are based on ^3H concentration of $1.2 \times 10^{-2} \mu\text{Ci}/\text{m}^3$ and ^{14}C concentration of $4.5 \times 10^{-5} \mu\text{Ci}/\text{m}^3$. See text for equations used to calculate doses.

The approximations and assumptions detailed above were chosen to provide a probable upper-bound estimate of EDE that would be received on an annual basis. Comparisons between these results and on-site ambient monitoring data were made to check the analysis. The relationship between emissions and ambient concentration can be described by an atmospheric dispersion term, A_D (Ci/y per pCi/m^3), defined as

$$A_D = \frac{Q}{C} \quad (\text{H.17})$$

where

- Q = annual ^3H point source emission rate (Ci/year),
 C = annual ambient ^3H concentration (pCi/m^3).

To calculate the potential SWSA 6 A_D value, area emission rates for the SWSA 6 waste disposal units were converted to an equivalent point source emission rate. Equivalent point source emissions can be calculated by

$$Q = JA \left(3600 \frac{\text{s}}{\text{h}} \right) \left(8760 \frac{\text{h}}{\text{year}} \right), \quad (\text{H.18})$$

where

J = vapor flux out of a waste disposal unit ($\text{Ci}/\text{m}^2 \cdot \text{s}$),

A = waste disposal unit surface area (m^2).

Table H.5 shows J , A , and the calculated Q value for each disposal unit at SWSA 6. If all the sources are approximated as a single point in the SWSA 6 facility, then the total emission rate is equal to 1630 Ci/year. Since the maximum average annual concentration predicted by the ISCLT3 model is 1.2×10^4 pCi/ m^3 , A_D was calculated using Eq. (H.18) and found to be 0.14 Ci/year per pCi/ m^3 (Table H.6).

For on-site calculations, almost all of the ^3H emissions on the Oak Ridge Reservation (ORR) occur from the Building 3039 stack at Oak Ridge National Laboratory (ORNL) (MMES 1991, 1992, and 1993). The nearest ambient ^3H monitor is located about 1100 m (3600 ft) to the southwest of this stack. Table H.6 shows emission rate and monitored ambient concentration values for 1990 through 1992 and the calculated A_D values. The value of A_D ranges between 94 and 364 Ci/year per pCi/ m^3 for these years.

The A_D values calculated using on-site data are approximately one order of magnitude higher than the A_D calculated from SWSA 6 modeling results. A lower A_D value indicates that atmospheric dispersion is less and that overall impact from the atmospheric pathway would be higher. Two reasons that the A_D value calculated for SWSA 6 is lower are that the on-site source is elevated and that the distance between the stack and monitor is about 1100 m. For the SWSA 6 modeling analysis, the source is at ground level and receptors were placed at a downwind distance very close to the site. Accounting for these differences would probably bring the modeled A_D value within about two orders of magnitude (Turner 1994) of the on-site A_D values. This two-order-magnitude difference is attributable to the conservatism employed in the analysis. Although there is no monitoring for ^{14}C on the ORR, results would probably be similar to those discussed above for ^3H . Therefore, the upper-bound assumptions in this analysis support the decision discussed in Sect. 3.2.1 to dismiss further analysis of atmospheric releases. Emissions of volatile radionuclides, as well as nonvolatiles, would lead to maximum off-site doses that are well below the 10 mrem per year performance standard.

Table H.5. Diffusive flux, modeled waste disposal surface area, and calculated potential atmospheric emission rates of ^3H at the SWSA 6 disposal units

Disposal unit	Diffusive flux (Ci/m ² ·s)	Modeled waste disposal unit surface area (m ²)	Point source emission rate (Ci/year ^a)
IWMF	1.86×10^{-8}	2.79×10^3	1.63×10^3
Tumulus I	1.84×10^{-14}	2.65×10^2	2.69×10^{-4}
Tumulus II	1.28×10^{-14}	4.65×10^2	1.88×10^{-4}
Low-range silos, north (zone 8)	3.07×10^{-13}	1.16×10^3	1.13×10^{-2}
Low-range silos, south (zone 30)	4.46×10^{-14}	3.48×10^3	4.89×10^{-3}
High-range silos	1.18×10^{-13}	3.25×10^3	1.21×10^{-2}

^aPoint source emission rates were calculated as the product of the diffusive flux, modeled waste disposal surface area, and appropriate unit conversion factors.

Table H.6. Emission rate, ambient concentrations, and calculated atmospheric dispersion term values for the SWSA 6 disposal units and the Oak Ridge Reservation

Location	Emission rate, Q (Ci/year)	Av. annual ambient atmospheric concentrations, C (pCi/m ³)	Atmospheric dispersion term, A_D (Ci/year per pCi/m ³) ^a
SWSA 6	1,630 ^b	12,000 ^c	0.14
Oak Ridge Reservation, 1990	12,000 ^d	33 ^e	364
Oak Ridge Reservation, 1991	16,000 ^d	4,700 ^e	94
Oak Ridge Reservation, 1992	2,140 ^d	2,250 ^e	113

^aThe atmospheric dispersion term is calculated as the ratio of the emission rate to the ambient concentration.

^bThe emission rate for SWSA 6 was calculated as the sum of the products of the radionuclide diffusive flux and the surface area of each disposal unit.

^cAmbient atmospheric concentrations were calculated using the Industrial Source Complex—Version 3 Long-Term Air Dispersion Model (ISCLT3) (EPA 1995). The maximum annual concentration was calculated to occur at a receptor located to the southwest of the IMWF.

^dAlmost all of the ^3H emissions for the ORR emanate from the Building 3039 stack located at ORNL (MMES 1991, 1992, and 1993).

^eAmbient atmospheric ^3H concentrations are measured at a monitor located at the perimeter of ORNL. This monitor is located about 1100 m to the southwest of the Building 3039 stack (MMES 1991, 1992, and 1993).

REFERENCES

- Copper, C. D., and F. C. Alley 1986. *Air Pollution Control: A Design Approach*, Waveland Press, Prospect Heights, Ill.
- DOE (U.S. Department of Energy) 1988. *Internal Dose Conversion Factors for Calculation of Dose to the Public*, DOE/EH-0071, Washington, D.C.
- EPA (U.S. Environmental Protection Agency) 1988. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion*, Federal Guidance Report No. 11, EPA 520/1-88-020, Office of Radiation Programs, Washington, D.C.
- EPA (U.S. Environmental Protection Agency) 1995. *User's Guide for the Industrial Source Complex (ISC3) Dispersion Models*, Vol. 1, *User's Instructions*, EPA-454/B-95-003a, Research Triangle, N.C.
- EPA (U.S. Environmental Protection Agency) 1992. *User's Guide for CAN-PC Version 1.0, Air and Radiation*, 402-B-92-001, Las Vegas, Nev.
- ICRP (International Committee on Radiation Protection) 1975. *International Commission on Radiological Protection, Task Group Report on Reference Man*, ICRP Publication No. 23, Pergamon, New York.
- MMES (Martin Marietta Energy Systems, Inc.) 1991. *Oak Ridge Reservation Environmental Report for 1990*, Vol. 2, *Data Presentation*, ES/ESH-18/V2, Martin Marietta Energy Systems, Inc., Oak Ridge, Tenn.
- MMES (Martin Marietta Energy Systems, Inc.) 1992. *Oak Ridge Reservation Environmental Report for 1991*, Vol. 1, *Narrative, Summary, and Conclusions*, ES/ESH-22/V1, Martin Marietta Energy Systems, Inc., Oak Ridge, Tenn.
- MMES (Martin Marietta Energy Systems, Inc.) 1993. *Oak Ridge Reservation Environmental Report for 1992*, Vol. 1, *Narrative*, ES/ESH-31/V1, Martin Marietta Energy Systems, Inc., Oak Ridge, Tenn.
- MMES (Martin Marietta Energy Systems, Inc.), EG&G Idaho, Inc., and Westinghouse Savannah River Company 1994. *Radiological Performance Assessment for the E-Area Vaults Disposal Facility*, WSRC-RP-94-218, Westinghouse Savannah River Company, Aiken, S.C.
- Napier et al. 1988. *GENII—The Hanford Environmental Radiation Dosimetry Software System*, Vol. 1, *Conceptual Representation*, Battelle, Pacific Northwest Laboratory, Hanford, Wash.
- NRC (U.S. Nuclear Regulatory Commission) 1977. *Regulatory Guide 1.109: Calculation of Annual Doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I*, Washington, D.C.

Turner, D. B. 1994. *Workbook of Atmospheric Dispersion Estimates: An Introduction to Dispersion Modeling*, 2nd ed., Lewis Publishers, Boca Raton, Fla.

Weast, R. C., and M. J. Astle, eds. 1981. *CRC Handbook of Chemistry and Physics*, CRC Press, Boca Raton, Fla.

Appendix I

SENSITIVITY AND UNCERTAINTY ANALYSIS FOR THE SWSA 6 PERFORMANCE ASSESSMENT

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Appendix I

SENSITIVITY AND UNCERTAINTY ANALYSIS FOR THE SWSA 6 PERFORMANCE ASSESSMENT

I.1 INTRODUCTION

I.1.1 Background

Calculated radionuclide concentrations presented in this performance assessment are determined by the Composite Transport Model (CTM), a linked series of numerical models, which approximates the transport of radionuclides from source units to drinking water pathways at SWSA 6. Because the maximum allowable dose of radiation from a given radionuclide in drinking water corresponds directly to a maximum concentration of that radionuclide, the uncertainty associated with such a dose is also directly proportional to uncertainties in calculated concentrations. Such uncertainties are assumed to come from two basic sources. The first of these is that, although the model is an accurate representation of the contaminant transport process, the physical parameters are never exactly known. For example, uncertainties arise due to the fact that both the type and quantity of past disposals of radionuclides at various SWSA 6 locations were generally poorly documented, making it necessary to conduct the analysis using reported inventory values. Other uncertainties exist because most probable or "average" values were used for many of the associated model input variables that vary significantly with space and/or time (e.g., material properties and environmental transport parameters). The second major source of uncertainty comes from the fact that the composite model may not be sufficiently specific to adequately describe the current site and its details and, at the same time, may not be sufficiently robust to describe the site for thousands, or even millions, of years.

Consequently, the sensitivity of the models used to analyze the performance of SWSA 6 and the uncertainty associated with the calculated environmental concentrations presented in Sect. 5.1 are important considerations in evaluating compliance of SWSA 6 with the performance objectives in DOE O 5820.2A and the water resource protection performance objectives described in Sect. 2.5.3. Hence, in addition to comparing predicted environmental concentrations to some theoretical limit, consideration must also be given to these uncertainties to ensure that these objectives will be met. The requirements necessary for compliance with these objectives, based on deterministic analyses for a number of possible limiting scenarios (e.g., groundwater well and surface water), are presented (see Sect. 5.1.2.2). Depending on future regulatory policy, the actual limiting scenario is subject to change. Therefore, rather than provide an absolute value for the probability that a given

concentration associated with a particular scenario results in compliance with the performance objectives of DOE O 5820.2A, the intent of this uncertainty analysis is to give the user a means of evaluating the nature and magnitude of uncertainties associated with the deterministic calculations so that performance objectives can be met.

From the initial list of 63 radionuclides considered in the PA analysis of SWSA 6 (Table 4.2), the following radionuclides, chosen to encompass a diverse range of environmental transport characteristics (e.g., combinations of K_d , half-life, and solubility), were analyzed to illustrate the effects that these uncertainties have on predicted environmental concentrations:

Table I.1. Radionuclides selected for SWSA 6 performance assessment uncertainty analysis

^3H	^{232}Th
^{14}C	^{233}U
^{90}Sr	^{238}U
^{99}Tc	^{239}Pu
^{137}Cs	^{241}Am

For purposes of this discussion, the models and databases related to waste disposals, performance of engineered systems, and release and transport of radionuclides through the environment to assumed human receptor locations are treated separately from the models for estimating dose per unit concentration of radionuclides at assumed receptor locations. This separation of the treatment of uncertainties can be justified on the grounds that predictions of radionuclide concentrations in the environment at assumed human receptor locations are based on analyses of physical systems, whereas the analysis of doses per unit concentration of radionuclides in the environment is based on largely hypothetical constructs that do not depend on the physical disposal system.

I.1.2 Definitions

I.1.2.1 Uncertainty/Entropy Relation

The uncertainty (U) associated with a radionuclide concentration, C , calculated using the CTM, is directly associated with the probability that $C \leq C^*$ (P_c), where C^* is some reference concentration. If $P_c = 0.5$, then no confident conclusion can be drawn as to whether or not $C \leq C^*$; there is an "even chance" that $C \leq C^*$ or $C > C^*$, and the analysis supplies no statement as to whether C does or does not exceed C^* . A quantitative measure of uncertainty, borrowed from information theory, is the computable quantity (the "entropy"):

$$U = -[P_c \cdot \log_2(P_c) + (1-P_c) \cdot \log_2(1-P_c)] \quad (I.1)$$

If $P_c = 0.5$, the uncertainty is at its maximum: $U = 1$. If $P_c = 0$ or $P_c = 1$, then $U = 0$. Thus, any feature of the SWSA 6 analysis that has the effect of driving P_c nearer to 0.5 is a feature that contributes to the uncertainty of the assertion that $C \leq C^*$. With this approach, the uncertainty in the judgment is scaled between 0 and 1. If $P_c = 1$, then the model predicts that $C \leq C^*$ with no uncertainty ($U = 0$). If $P_c = 0$, then the model predicts that $C > C^*$, again with no uncertainty. But, if $P = 0.5$, essentially no definite conclusion can be made as to whether C is less than, equal to, or greater than C^* .

It is assumed that the actual concentration, C , of a given radionuclide at a given point in time and location is a random-valued quantity. The maximum value of the (unknown) concentration over all time, starting at 1988, is assumed to be a random variable determined by a plethora of random and planned events, including inventory, weather and climate, concrete and steel durability, and future geologic and hydrologic events. Since some radionuclides (e.g., ^{238}U and ^{232}Th) have half-lives in the billions of years, some of the modeled predictions occur on a geological time scale. Hence, the random events that determine actual concentrations are virtually limitless in scope, and any model must be put in proper perspective with some evaluation of accompanying uncertainty beyond routine Latin hypercube (LHC) analysis.

I.1.2.2 Parametric or Objective Uncertainty

Engineering calculations such as the PA of SWSA 6 generally require extensive amounts of information to describe a physical quantity or process. Consequently, the uncertainties have a potentially significant impact on both the validity and usefulness of the analysis results. For example, considerable uncertainty may be associated with both the types and amounts of radionuclides present at a given disposal unit. Another obvious form of uncertainty associated with these calculations is the fact that inputs to numerical contaminant transport models, such as physical or environmental parameters, rarely are represented by a single value. Material properties generally vary with temperature, pressure, or time. Similarly environmental transport parameters are seldom homogeneous across a site such as SWSA 6. Often values for these quantities can vary significantly even over small distances at a site. In theory, most numerical models allow for the input of multiple values for these quantities; however, many of these quantities can only be determined through costly, experimental measurements. Furthermore, it is essentially impossible to estimate future values for parameters, such as annual rainfall, that vary with time. Therefore, a single "representative" or "mean" value is generally used to quantify these parameters in the modeling process.

Generally the magnitude of these parametric objective uncertainties can be quantified using random sampling techniques such as Monte Carlo analysis in which each input parameter is characterized by a probability density function (PDF), rather than a single, deterministic value. A series of model input data sets are then constructed by randomly sampling from these distributions. Model

iterations are then performed using these input sets to obtain distributions of desired output variable such as contaminant flux rates or concentrations at given receptor points. These resulting distributions can be used to determine the probability that a given output variable will fall within a certain range. Obviously with this method, the accuracy of the probabilities obtained from these distributions is directly proportional to the number of random samples. Even with the power of modern computers, for complicated numerical models this can lead to significant amounts of computational time and mass storage requirements. Therefore, a more desirable approach is to use systematic sampling techniques such as LHC sampling. LHC is similar to Monte Carlo analysis, but rather than randomly sampling from the input parameter distributions, the distributions are first divided into N equal probability intervals (where N is the number of model iterations to be performed). This ensures that the entire range of each parameter will be evenly sampled and results in the same accuracy as simple random sampling techniques while requiring about ten times fewer samples (Gardner, Rödger, and Bergström 1983).

The list of input variables required to perform numerical calculations such as those conducted in this PA can be extensive. For many of these parameters, determination of representative input distributions may be a difficult or time-consuming task. Thus, just as techniques such as LHC analysis can be used to reduce the number of model iteration required in an uncertainty analysis, it is also desirable to limit the number of parameters considered in the analysis to only those that have the greatest impact on desired output. For such cases, common practice is to perform sensitivity analyses in which the response of selected output quantities to perturbations of model input parameters are calculated. These results can then be used to identify the parameters with the greatest influence on model output, thereby reducing the number of variables that must be considered for the uncertainty analysis.

I.1.2.3 Subjective Uncertainty

The uncertainty due to parameter variance is only one component of the total uncertainty associated with the environmental transport modeling of radionuclides at SWSA 6. Additional uncertainties exist because the models and assumptions used in such analyses are only approximations of an actual system and may be inadequate for making long-term predictions. Quantitative values for these uncertainties are not explicitly calculated or measured but subjectively estimated based on experience. In attempting to factor into the analysis subjective assessments, these perceived sources of uncertainty should be kept separate. That is to say, it is useful to draw a distinction between "objective uncertainty" and "subjective uncertainty," where the former is associated solely with the propagation of parameter variance through a model or series of models. The objective uncertainty is expressed by the PDFs derived from the LHC simulations. These PDFs do not account for the uncertainty associated with a judgment about the actual "validity" of the model. This is a subjective assessment best made by the model developer, or expert, who can express his or her confidence in the fundamental validity of each model segment. If expressed appropriately, this subjective judgment of confidence can then be incorporated into the uncertainty analysis, thus

modifying the objective LHC PDFs. The effect of incorporating subjectivity in this fashion is to smear the objective distributions somewhat, hence increasing the uncertainty associated with model calculations. To show where judgments of validity in a model may arise, it is useful to break down each component of the transport model into the elements listed in Table I.2. Sources of subjective uncertainty can occur for any of the reasons noted. All of these hierarchical elements may contribute to a greater or lesser extent to an overall subjective judgment about the validity of the model.

Table I.2. Sources of subjective uncertainty in the transport model

Element of modeling process	Course of uncertainty
Conceptual model	Incomplete understanding of all aspects of the transport process or uncertainty in underlying assumptions of the model
Mathematics/physics	Failure to adequately capture or express conceptual understanding in mathematical language
Numerical	Failure to formulate the mathematics in usable algorithms
Computer model	Failure to eliminate errors in resulting computer codes
Interpretation of results	Difficulty in interpreting results in terms of the conceptual model

I.1.3 Methods for Quantifying Uncertainties Associated with Calculated Radionuclide Concentrations at SWSA 6

I.1.3.1 Approach for Quantifying Parametric or Objective Uncertainties

To demonstrate the effect of parametric uncertainties associated with calculated environmental contaminant concentrations at SWSA 6, a parametric uncertainty analysis was performed on the ten radionuclides listed in Table I.1. For this analysis, limiting inventories based on maximum groundwater concentrations at 100-m compliance points (see Sect. 5.1.2.2, Table 5.27) were used for the Interim Waste Management Facility (IWMF) and were assumed to contain no uncertainty, as indicated in Sect. I.2. Inventories for the remaining disposal sites were subject to uncertainty and are reported in detail in Sect. 6 of Appendix A.

As described in Sect. I.1.2.2, the effects of the parametric uncertainties associated with a given model are typically quantified by applying sampling techniques such as LHC analysis to determine distributions for selected output quantities. However, rather than a single model component, the CTM used in the performance assessment of SWSA 6 consists of six components (or stages):

- determination of initial mass (inventory),
- determination of representative precipitation at SWSA 6 for tens of thousands of years,
- prediction of release rates for each site and each radionuclide,
- prediction of transport through the saturated groundwater system,
- prediction of transport to surface water from the shallow subsurface and groundwater discharge, and
- prediction of doses based on future contaminant concentrations and presumed future consumer behavior.

Each of these components introduces its own, unique uncertainties, which contribute to the overall parametric uncertainty associated with model calculations. Thus, to quantify this total parametric uncertainty associated with CTM calculations, a coupled sampling procedure, which both propagated uncertainties due to the previous model stage's calculations and introduced new uncertainty associated with the current model stage, was conducted independently for each of the radionuclides in Table I.1. This procedure, shown in Fig. I.1., is described in detail below.

The first step was to characterize the disposal of each selected radionuclide at each site within SWSA 6 by a probability distribution (Sect. I.2). Next, a similar process was conducted in which probability distributions were assigned to critical input parameters of the SOURCE and Unified Transport Model (UTM) models (See Sect. I.4 and Sect. I.3). LHC sampling techniques were then used to generate 50 SOURCE model input data sets from these distributions. These data sets were then processed using the SOURCE model, resulting in 50 data sets of contaminant fluxes and associated volumetric water fluxes in the vertical direction, as well as 50 data sets of contaminant fluxes and associated volumetric water fluxes in the lateral direction. Using these output data sets and the data sets obtained from LHC sampling of probability distributions assigned to critical input variables of the PADSIM/HOLSIM model (See Sect. I.5), 50 input data sets to the PADSIM/HOLSIM were created and input into PADSIM/HOLSIM, resulting in 50 data sets of contaminant fluxes to surface water and recharge. An analogous process was then performed where the 50 data sets of contaminant flux to recharge output from PADSIM/HOLSIM, along with the 50 data sets obtained from LHC sampling of distributions of critical input parameters to the U.S. Geological Survey's (USGS's) Method of Characteristics (MOC) groundwater model (See Sect. I.6), were used to create 50 input data sets for MOC. Using this input, 50 MOC iterations were performed resulting in 50 data sets of maximum groundwater concentrations at receptor locations and 50 data sets of contaminant fluxes to surface water. The latter were combined with the 50 data sets of contaminant fluxes to surface water calculated by PADSIM/HOLSIM to give 50 data sets of total contaminant flux to surface water.

Fig. I.1 Figure showing coupled sampling procedure.

I.1.3.2 Approach for Quantifying Subjective Uncertainties

Below are described two possible approaches for incorporating the subjective expert's opinion about whether a model (or model component) is "valid." The first approach (Sect. I.1.3.2.1) uses a very simple, but crude, weighting scheme that yields a lower bound on the probability that a calculated value for a physical quantity (such as contaminant concentration at a receptor location) is less than or equal to the actual value. This approach requires a subjective assessment from each expert in the form of a single number between 0 and 1 that attempts to assign a weight to the validity of his or her model component. The second approach (Sect. I.1.3.2.2) attempts to incorporate expert opinion in a more rigorous fashion but requires a more quantitative expression of the model's validity from the expert, as explained below. This method was used in the present analysis to illustrate the additional effects of subjective uncertainties associated with calculated concentrations for the following radionuclides: ^3H , ^{14}C , ^{90}Sr , ^{137}Cs , and ^{238}U .

I.1.3.2.1 Lower Bound Method for Quantifying Subjective Uncertainty

In the lower-bound approach, an expert assigned to a particular segment of the CTM is asked to provide a weight, w , with $0 \leq w \leq 1$, where the choice $w = 1$ means that his or her segment is (completely) valid in the sense that the expert has a significant degree of confidence in all elements of the model, as listed above. Conversely, the weight $w = 0$ means that the expert has no confidence that his or her segment is capable of representing the physical system, and is, therefore, not suitable for prediction. Thus, choosing w amounts to a subjective estimate of how relevant the model is in representing the physical system and making predictions. I.E., w is a measure of the degree of confidence that a model component is valid. The goal will be to incorporate the choice for w into the more standard "objective" uncertainty analysis of parameter variance based on LHC sampling. To this end, it is convenient to define w as the probability of validity:

$$w_i: \text{Pr}(\text{model segment } i \text{ is valid}) .$$

Each component in the model may also be assigned a validity weight, as follows:

- w_1 : Pr(initial mass approximation is valid)
- w_2 : Pr(hydrologic model is valid)
- w_3 : Pr(contaminant release model is valid)
- w_4 : Pr(transport to shallow subsurface and groundwater recharge model is valid)
- w_5 : Pr(groundwater transport model is valid)
- w_6 : Pr(consumer scenario is valid)
- w_7 : Pr(dose calculation is valid).

For the purpose of identifying the “weakest link” in the model, it is reasonable to take the overall validity as the minimum:

$$w: \text{Minimum } \{w_1, w_2, w_3, w_4, w_5, w_6, w_7\} .$$

This weight can now be combined with the computational uncertainty estimates derived from the LHC sampling simulations as follows.

Viewing C (contaminant concentration) as an example random variable, the event $C \leq C^*$ occurs with a certain probability. This event can evidently happen in two ways:

$$C \leq C^* \text{ and (model is valid)}$$

or

$$C \leq C^* \text{ and (model is not valid)} .$$

Let A and $\tilde{A}$ be events defined as follows:

$$A : \text{Model segment is valid}$$

$$\tilde{A} : \text{Model segment is not valid} .$$

Note that the definition $w = \Pr(A)$ implies $\Pr(\tilde{A}) = 1 - w$. Then the event can be related to the probability that the composite model actually predicts $C \leq C^*$ by the formula for total probability:

$$\Pr(C \leq C^*) = \Pr(C \leq C^* | A) \cdot \Pr(A) + \Pr(C \leq C^* | \tilde{A}) \cdot \Pr(\tilde{A}) \quad (\text{I.2})$$

or

$$\Pr(C \leq C^*) = \Pr(C \leq C^* | A) \cdot w + \Pr(C \leq C^* | \tilde{A}) \cdot (1 - w) . \quad (\text{I.3})$$

As a simple, common sense check, notice that if $w = 1$, then $\Pr(C \leq C^*) = \Pr(C \leq C^* | A)$, which is to say that the sought result $\Pr(C \leq C^*)$ is given wholly by $\Pr(C \leq C^* | A)$. That is, $\Pr(C \leq C^*)$ can be determined *computationally* from an analysis of the model. [This is precisely the assumption when parameter propagation studies are used solely to determine uncertainty (e.g., via LHC sampling simulations).] If $w = 0$, then $\Pr(C \leq C^*) = \Pr(C \leq C^* | \tilde{A})$, which is to say that the desired quantity should be determined somehow with no regard for the model. Finally, because it is always true that $0 \leq \Pr(C \leq C^* | \tilde{A}) \leq 1$, Eq. (I.3) implies the following:

$$\Pr(C \leq C^*|A) \cdot w \leq \Pr(C \leq C^*) \leq \Pr(C \leq C^*|\tilde{A}) \cdot w + 1 \cdot (1 - w).$$

The lower half of the inequality, $\Pr(C \leq C^*|A) \cdot w \leq \Pr(C \leq C^*)$, shows that an LHC sampling-derived value for $\Pr(C \leq C^*|A)$ and a subjective weight w , chosen for model validity, yield a lower bound for the probability that $C \leq C^*$.

I.1.3.2.2 Method for Quantifying Subjective Uncertainty Based on Treatment of Expert Opinion as a "Clue"

This section describes a more rigorous method in which an expert's opinion is treated as a clue that can be used to modify an objective probability distribution. This was the approach chosen for the current study.

To use subjective information to modify the LHC cumulative distribution functions (CDFs) using the rules of probability, an expert assigned to a particular model segment is queried as follows: "Under the assumption that the concentration predicted by the model is based on the true parameters, express your confidence in this value by drawing a confidence band around the predicted value." Using $\hat{C}$ and C to denote, respectively, the predicted and true contaminant concentrations associated with the model segment, one can then approximate the conditional probability distribution $P^{\text{Subj.}}(C|\hat{C})$ from this confidence band. (Below, the symbol P will refer to a PDF.) The modified distribution is then derived from the rule of total probability:

$$P(C) = \int P^{\text{Subj.}}(C|\hat{C})P^{\text{LHC}}(\hat{C})d\hat{C}, \quad (\text{I.4})$$

where $P^{\text{LHC}}(\hat{C})$ is the LHC sampling-derived distribution. The above integration has the effect of broadening the original $P^{\text{LHC}}(\hat{C})$, and, hence, of increasing the uncertainty that $C \leq C^*$. Eq. (I.4) shows how subjective information is used to modify the LHC sampling distribution in one model segment. In the propagation of probabilities through the daisy chain, subjective information can be incorporated at each stage. This process is described by the next three equations. Below, C_{in}^i represents the contaminant concentration entering the i -th stage, and C_{out}^i is the contaminant concentration predicted (or "output") from the i -th stage. The term C_{out}^i denotes the LHC sampling-derived ("objective") output probability from the i -th stage. Recall that the output concentration of one model component is used as the input for the next component, that is, $C_{\text{out}}^{i-1} = C_{\text{in}}^i$. This implies

$$P(C_{\text{in}}^i) = P(C_{\text{out}}^{i-1}). \quad (\text{I.5})$$

Now let $P^{LHC}(\hat{C}_{out}^i | C_{in}^i)$ denote the conditional probability distribution that expresses model uncertainty as estimated by the LHC sampling simulations for component i . Then the following relation shows how the contaminant PDF predicted by the previous model component, $P(C_{out}^{i-1}) [= P(C_{in}^i)]$, is affected by parameter variability in the present component:

$$P(\hat{C}_{out}^i) = \int P^{LHC}(\hat{C}_{out}^i | C_{in}^i) P(C_{in}^i) dC_{in}^i . \quad (I.6)$$

The next step is to modify the “objective” distribution, $P(C_{out}^i)$, using the expert's subjective assessment about the “validity” of model segment i . This information is expressed by the conditional PDF, $P^{Subj}(C_{out}^i | \hat{C}_{out}^i)$. This conditional distribution is constructed from the expert's opinion as follows: given the model prediction of $\hat{C}_{out}^i$ based on the LHC sampling simulation, a probability distribution (or confidence band) is drawn around this predicted value that expresses the expert's confidence in this value and, hence, in the validity of the model. When this is done, the following relation shows how the output (LHC sampling) distribution for the i -th segment, $P(\hat{C}_{out}^i)$, is modified by the subjective uncertainty to yield the new output distribution for the i -th segment $P(C_{out}^i)$:

$$P(C_{out}^i) = \int P^{Subj}(C_{out}^i | \hat{C}_{out}^i) P(\hat{C}_{out}^i) d\hat{C}_{out}^i . \quad (I.7)$$

This is merely a restatement of Eq. (I.4) for segment i .

The above equations define how the probability distribution of the i -th segment is derived from that of the previous segment. A sequence of such equations can then be linked together to obtain the probability distribution associated with the composite model, namely, the output distribution from the final segment, $P(C_{out}^N)$, where N is the total number of segments. The composite distribution is then used to compute the probability that $C \leq C^*$:

$$P_r(C \leq C^*) = \int_0^{C^*} P(C_{out}^N) dC_{out}^N , \quad (I.8)$$

C^* being the same reference concentration. From this result, one can compute the compliance uncertainty defined by the entropy relation [Eq. (I.1)]. It can be shown that the effect of incorporating subjective information into the analysis in this way almost always results in an increase in uncertainty as defined by Eq. (I.1). (Exceptions can be found if the subjective probability distribution is significantly skewed in one direction—for example, to convey a judgment that the LHC sampling predictions are extremely conservative.) This can be seen by noting that subjective information tends to “smooth” the objective (LHC sampling) distribution function, with the result that the probability “density” is distributed more evenly above and below the point C^* ; this drives $\Pr(C \leq C^*)$ toward 0.5 and, hence, U toward 1. In other words, if the LHC sampling-predicted probability, $\Pr(C \leq C^*)$, is

initially far from 0.5, subjective information will tend to force it closer to 0.5; as a consequence, the addition of subjective information can *increase* $\Pr(C \leq C^*)$ if it is initially below 0.5. The more important case, however, occurs when the LHC sampling analysis predicts $\Pr(C \leq C^*)$ between 0.5 and 1.0. Here, additional subjective information will tend to decrease $\Pr(C \leq C^*)$ and again push the uncertainty, U , closer to 1.

Under certain conditions, it is possible, and computationally convenient, to compute the overall (composite) objective and subjective probability distributions separately and then to combine these composite distributions using a relation similar to Eq. (I.4). That is, one first performs the LHC sampling simulation on the composite model without incorporating subjective information at each stage. The individual subjective distributions are combined separately to form an overall subjective distribution. This is then used to modify the composite LHC sampling distribution according to Eq. (I.4). The separate computation of the objective and subjective distributions can be justified if Eq. (I.6) and Eq. (I.7) are approximate convolution integrals. The latter assumption holds if the transport models are linear in the contaminant concentration (i.e., given that doubling the "input" contaminant concentration implies a doubling of the "output" concentration). Although a particular model segment may be highly nonlinear in the total flow field (e.g., groundwater transport), if the contaminant concentration itself represents a small perturbation of the total flow volume, it is reasonable to expect that the model will behave linearly in the contaminant concentration to first order. With this assumption and an appropriate normalization of the concentration scales, Eq. (I.6) and Eq. (I.7) can be expressed as convolution integrals. Then the composite probability distribution of an N -stage model may be written as

$$P_N = [P_{N-1}^{\text{LHC}} * P_{N-2}^{\text{Subj}}] * [P_{N-2}^{\text{LHC}} * P_{N-2}^{\text{Subj}}] * \dots * [P_1^{\text{LHC}} * P_1^{\text{Subj}}] , \quad (\text{I.9})$$

where $*$ denotes convolution and the subscript indicates model segment. Because the convolution operation is commutative, the order of the convolutions can be rearranged to yield

$$P_N = P_{\text{total}} = P_{\text{total}}^{\text{LHC}} * P_{\text{total}}^{\text{Subj}} , \quad (\text{I.10})$$

where

$$P_{\text{total}}^{\text{LHC}} = P_{N-1}^{\text{LHC}} * P_{N-2}^{\text{LHC}} * \dots * P_1^{\text{LHC}} \quad (\text{I.11})$$

and

$$P_{\text{total}}^{\text{Subj}} = P_{N-1}^{\text{Subj}} * P_{N-2}^{\text{Subj}} * \dots * P_1^{\text{Subj}} . \quad (\text{I.12})$$

Here, Eq. (I.11) is computed by an LHC sampling simulation applied to the entire composite model, and Eq. (I.12) shows that the total subjective distribution can be estimated by convoluting the individual (properly normalized) subjective distributions of the separate model segments.

I.1.3.2.3 Computer Algorithm for Incorporation of Subjective Uncertainty

The methods presented in Sect. I.1.3.2.2 for incorporating subjective uncertainties with the parametric uncertainties associated with the LHC analyses of CTM are implemented through the use of a computer program. The code consists of about 300 lines written in FORTRAN. It is compiled and run on an HP 9000, Series 700, workstation. The code uses the following approximation to Eq.(I.4) to combine subjective information with modeled, "objective" results:

$$P(C_i) = P^{Subj}(C_i|\hat{C}_j) * P_j^{LHC}(\hat{C}) \quad , \quad (I.13)$$

where

$$\begin{aligned} P(C_i) &= \text{probability that the actual concentration is in interval } i; \\ P^{Subj}(C_i|\hat{C}_j) &= \text{subjective probability for a given model segment that the} \\ &\quad \text{actual concentration, } C, \text{ is in interval } i \text{ given the modeled} \\ &\quad \text{concentration, } \hat{C} \text{ is in interval } j; \\ P_j^{LHC}(\hat{C}) &= \text{probability that the modeled concentration, } \hat{C}, \text{ is in a given} \\ &\quad \text{interval, } j \text{ (originally this probability corresponds to the "objective",} \\ &\quad \text{LHC analysis results).} \end{aligned}$$

The following are brief descriptions of the steps used by the computer code.

Step 1

A concentration (or flux) range that encompasses all values generated by the LHC analysis for a given nuclide, as well as the value associated with the 4mrem/year dose limit, C^{lim} , is determined. This range is then divided into n unit-width intervals. While the actual number of intervals, n , is nuclide-specific, the values used in this analysis ranged from 20 to 60.

Step 2

For each of the five model segments, subjective probabilities are assigned for the n intervals, $P^{Subj}(C_i|\hat{C}_j)$. The code allows input of 11 subjective values, corresponding to $P^{Subj}(C_i|\hat{C}_{i-5})$ to $P^{Subj}(C_i|\hat{C}_{i+5})$. These probabilities are based on distributions provided by experts on the particular CTM segment. A "rest-of-the-world" probability, uniformly distributed over the remaining $(n-11)$ intervals, is also input.

Step 3

Once the intervals and subjective probabilities have been stored, initial values for $P_j^{LHC}(\hat{C})$ based on the objective distributions obtained from the LHC analysis, are generated. These probabilities are simply the ratio of the number of objective values that fall within each interval and the total number of objective values (i.e., 50).

Step 4

Using the estimates for $P_j^{LHC}(\hat{C})$, and the subjective probabilities for the first model segment, values for $P(C_i)$ are calculated for each interval using Eq. (H.12).

Step 5

The previous probabilities, $P_j^{LHC}(\hat{C})$, are then replaced by the corresponding values of $P(C_i)$ calculated in Step 4.

Step 6

Using the updated values for $P_j^{LHC}(\hat{C})$ and subjective probabilities $P^{Subj}(C_i|\hat{C}_j)$ for the next model segment, Steps 4 and 5 are repeated for each of the remaining model segments.

Steps 4 and 5 are then repeated for each of the remaining model segments. These values are then used with the subjective estimates for the next model segment and Step 4 is repeated.

I.1.3.2.4 Subjective probability distributions

The subjective probability distributions for selected radionuclides (^3H , ^{14}C , ^{90}Sr , ^{137}Cs , and ^{238}U), which required that experts on each component of the CTM estimate the distribution of actual values relative to predicted concentrations (or fluxes) of value C , are summarized in Table I.3. Using the computer code described in Sect. I.1.3.2.3, the subjective distributions associated with contaminant inventory, UTM, the SOURCE1 and SOURCE2 models, and the PADSIM and HOLSIM were convoluted with the PADSIM and HOLSIM output distributions obtained from the parametric uncertainty analysis (Sect. 4.6.1.3) to obtain modified distributions of total concentration in surface water due to flux from the shallow subsurface. Similarly, by convoluting the subjective distributions associated with contaminant inventory, UTM, the SOURCE1 and SOURCE2 models, PADSIM and HOLSIM, and MOC, with the MOC output distributions obtained from the parametric uncertainty analysis, modified distributions (which reflected both subjective and parametric uncertainties) of: (1) contaminant concentration in groundwater, and (2) total contaminant concentration in surface water due to flux from groundwater were also obtained. Finally, distributions for total contaminant concentration in surface water were generated by combining the modified distributions of contaminant flux from shallow subsurface and flux from groundwater described above.

Table I.3. Description of subjective probability distributions for each segment of the Composite Transport Model

CTM model segment	Nuclide	Assumed distribution	Comments
Inventory	^3H	Uniform	50% of actual values within $\pm 50\%$ of predicted value
Inventory	^{14}C	Uniform	20% of actual values within $\pm 50\%$ of predicted value
Inventory	^{90}Sr	Uniform	60% of actual values within $\pm 100\%$ of predicted value
Inventory	^{137}Cs	Uniform	80% of actual values within $\pm 25\%$ of predicted value
Inventory	^{238}U	Uniform	40% of actual values within $\pm 35\%$ of predicted value
UTM recharge values	All	Uniform	90% of actual values within $\pm 20\%$ of predicted value
SOURCE model	^3H , ^{14}C , ^{90}Sr , ^{137}Cs	Lognormal	Mean equal to modeled concentration; standard deviation of 0.9
SOURCE model	^{238}U	Lognormal	Mean equal to modeled concentration; standard deviation of 1.5
PADSIM/HOLSIM model	^3H , ^{14}C , ^{90}Sr , ^{137}Cs	Uniform	90% of actual values within ± 1 order of magnitude of predicted values
PADSIM/HOLSIM model	^{238}U	Uniform	90% of actual values within ± 2 orders of magnitude of predicted values
USGS MOC model	^3H , ^{14}C	Lognormal	90% of actual values within ± 1 order of magnitude of predicted values
USGS MOC model	^{90}Sr , ^{137}Cs	Lognormal	90% of actual values within ± 2 orders of magnitude of predicted values
USGS MOC model	^{238}U	Loguniform	Loguniform over ± 5 orders of magnitude of predicted values

Acronyms: CTM = Composite Transport Model, UTM = Unified Transport Model, USGS = U.S. Geological Survey, MOC = Method of Characteristics.

I.2 UNCERTAINTIES ASSOCIATED WITH INVENTORIES

Appendix A of this report presents results from statistical analyses on the reported inventories at non-IWMF SWSA 6 disposal sites. The parametric uncertainty analysis was conducted using the reported inventories and the corresponding statistical parameters determined from the analysis in Appendix A (Tables A.16 - A.27).

The performance of SWSA 6 with respect to transport of radioactive contaminants in groundwater was initially evaluated by calculating limiting inventories for IWMF based on maximum groundwater concentrations at 100-m receptor locations (Fig. 4.15). The parametric uncertainty analysis was conducted using these limiting inventories for the IWMF facility. No uncertainty is assumed to exist in these values because it is assumed that quality assurance procedures implemented at the IWMF would guarantee that future disposals are accurately reported. Uncertainties associated with previous disposals at IWMF are discussed in Sect. 6 of Appendix A.

After completion of the parametric uncertainty analysis, it was realized that the associated groundwaters between the 100-m buffer zone and White Oak Lake in the vicinity of IWMF would certainly be protected to the same extent as the groundwaters inside the 100-m buffer zone. Therefore, the buffer zone for the various SWSA 6 disposal units was expanded (Fig. 4.16); and, revised limiting inventories for IWMF based on maximum groundwater concentrations at this expanded buffer zone were determined. However, due to lack of sufficient time and funding, it was not possible to redo the parametric uncertainty analysis using these revised IWMF inventories.

I.3 UNCERTAINTIES ASSOCIATED WITH UTM

Uncertainty analysis on UTM using LHC sampling is not conducted. a number of sensitivity analyses showing the influence of inputs on water budgets and chemical transport have been conducted with UTM (Luxmoore, Stolzy, and Holdeman 1976; Begovich and Luxmoore 1979; Sharma and Luxmoore 1979). The heterogeneity of plant physiological variables and soil physical properties have complex interacting effects that do not need to be included in this report. Such heterogeneity causes the quantity of water interacting with waste units to vary. Water budget calculations are performed for dry, average, and wet years (Table 4.14), and in general higher rainfall results in relatively more lateral flow and has essentially no effect on recharge. In the programming sequence, recharge is calculated first and this results in recharge being limited by hydraulic conductivity and not the supply of water. Thus, recharge is insensitive to variation in rainfall whereas lateral flow is highly sensitive to rainfall amount.

More or less lateral flow interacting with waste units results in lower or higher concentrations of nuclides entering surface water. Because the surface water flow from SWSA 6 is diluted with surface runoff from a perennial stream, the consequence of uncertainty in water budget components calculated by UTM is unimportant and is not included in the uncertainty analysis.

I.4 UNCERTAINTIES ASSOCIATED WITH SOURCE MODEL

The SOURCE1 and SOURCE2 computer codes require more than 100 input parameters for a performance simulation. However, sensitivity analyses can be used to reduce the number of parameters to be considered in uncertainty analyses. For conditions expected at Oak Ridge, previous studies (Icenhour and Tharp 1995; Icenhour, Tharp, and Miller 1997) have summarized the sensitive parameters for SOURCE1 and SOURCE2. These analyses were performed with the use of the PRISM computer code (Gardner, Røjder, and Bergström 1983). The sensitive parameters and associated values are shown in Table I.4, Table I.5, Table I.6, and Table I.7 provide information by nuclide for the input parameters that are radionuclide-specific.

Several parameters that related to the waste (or waste form) were identified as sensitive, including waste density, distribution coefficient, moisture content, radionuclide diffusion coefficient in concrete, radionuclide solubility, and radionuclide inventory. For the mechanism of sulfate attack, the sulfate diffusion coefficient and sulfate concentrations were identified as sensitive parameters. The water seepage parameters of soil hydraulic conductivity, concrete hydraulic conductivity, and disposal unit area were also sensitive. Finally, soil density; corrosion times for metals in containers, liners, and wells; and the initial functionality of the tumulus-type concrete pad and leachate collection systems were sensitive parameters.

The parameter distributions indicated in Table I.4, Table I.5, Table I.6, and Table I.7 were used in conjunction with the PRISM computer code to create 50 input data sets for each radionuclide in each disposal unit. These input data sets were then used in either SOURCE1 or SOURCE2 simulations to obtain 50 sets of computer code outputs. These outputs then served as part of the input parameters for PADSIM or HOLSIM.

I.5 UNCERTAINTIES ASSOCIATED WITH SHALLOW SUBSURFACE MODELS

The PADSIM and HOLSIM codes for simulation of chemical transport in the shallow subsurface contain five variables that are used with LHC sampling for uncertainty analysis. These variables are:

- bulk density,
- chemical adsorption coefficient (K_d) (Table I.8),
- disposal unit area (Table I.9),
- distance to groundwater (Table I.10), and
- distance to stream channel (Table I.11).

The regolith of the shallow subsurface transport path at SWSA 6 has a mean bulk density of 1.35 g/cm³ (Luxmoore 1982). A normal frequency distribution is assumed with a standard deviation,

Table I.4 SOURCE1 and SOURCE2 parameters and values included in the uncertainty analysis

Parameter	SOURCE1 ^a	SOURCE2 ^a	Mean	Standard deviation	Distribution	Minimum	Maximum
Earthen cover density, g/cm ³	Yes	Yes	1.76		Uniform	1.50	2.00
Waste density, g/cm ³	Yes	Yes	1.76		Uniform	1.00	2.50
Relative saturation of waste, unitless	Yes	Yes	0.35		Triangular	0.15	0.55
SO ₄ ²⁻ diffusion coefficient in concrete, m ² /s	Yes	Yes	1.06×10^{-12}		Triangular	1.06×10^{-12}	1.06×10^{-10}
SO ₄ ²⁻ concentration (inside unit), mol/L	Yes	Yes	2.62×10^{-4}	1.30×10^{-5}	Normal	2.23×10^{-4}	3.01×10^{-4}
SO ₄ ²⁻ concentration (outside unit), mol/L	Yes	Yes	2.62×10^{-4}	1.30×10^{-5}	Normal	2.23×10^{-4}	3.01×10^{-4}
Pad initial functionality fraction (IWMF), unitless	Yes	No	0.8		Uniform	0.6	1
Pad initial functionality fraction (TUM I & TUM II), unitless	Yes	No	0.5		Uniform	0.3	1
Disposal unit area, m ²	Yes	Yes	<i>b</i>	<i>b</i>	<i>b</i>	<i>b</i>	<i>b</i>
Steel-vault failure time, year	Yes	No	60		Uniform	30	90
Epoxy-coating failure time, year	Yes	Yes	20		Uniform	10	30
Steel-liner failure time, year	No	Yes	60		Uniform	30	90
Steel-well failure time, year	No	Yes	75		Uniform	25	125
Saturated hydraulic conductivity for recharge under unit (IWMF), cm/s	Yes	No	1.90×10^{-5}	1.90×10^{-5}	Lognormal	1.00×10^{-1}	1.90×10^{-2}
Saturated hydraulic conductivity for recharge under unit (TUM I & TUM II), cm/s	Yes	No	1.50×10^{-5}	1.50×10^{-5}	Lognormal	1.00×10^{-1}	1.50×10^{-2}
Saturated hydraulic conductivity for recharge under unit (SOURCE2), cm/s	No	Yes	5.80×10^{-7}	5.80×10^{-7}	Lognormal	1.00×10^{-11}	5.80×10^{-4}
Saturated hydraulic conductivity in concrete, cm/s	Yes	Yes	1.00×10^{-10}		Triangular	1.00×10^{-11}	1.00×10^{-9}
Distribution coefficient in waste, mL/g	Yes	Yes	<i>c</i>	<i>c</i>	<i>c</i>	<i>c</i>	<i>c</i>
Diffusion coefficient of nuclide in concrete, m ² /s	Yes	Yes	<i>c</i>	<i>c</i>	<i>c</i>	<i>c</i>	<i>c</i>
Solubility of nuclide, mol/L	Yes	Yes	<i>c</i>	<i>c</i>	<i>c</i>	<i>c</i>	<i>c</i>
Inventory, g/unit	Yes	Yes	<i>d</i>	<i>d</i>	<i>d</i>	<i>d</i>	<i>d</i>

Acronyms: IWMF = Interim Waste Management Facility, TUM = Tumulus.

^a"Yes" indicates that the parameter is applicable to this computer code (i.e., SOURCE1 or SOURCE 2).

^bValues for this parameter are unit-specific and are given in Table I.9.

^cValues for this parameter are nuclide-specific and are given in Table I.5, Table I.6, and Table I.7.

^dValues for this parameter are nuclide- and unit-specific and are given in Appendix A.

Table I.5 Uncertainty values for distribution coefficients used in the SOURCE1 and SOURCE2 computer programs

Nuclide	Distribution coefficient in waste (mL/g)			Distribution
	Probable	Low	High	
³ H	1.99×10^{-1}	0.00×10^0	1.99×10^0	Triangular
¹⁴ C	1.09×10^0	0.00×10^0	1.09×10^1	Triangular
⁹⁰ Sr	8.75×10^0	8.75×10^{-1}	8.75×10^1	Triangular
⁹⁹ Tc	3.18×10^0	0.00×10^0	3.18×10^1	Triangular
¹³⁷ Cs	1.99×10^1	1.99×10^0	1.99×10^2	Triangular
²³² Th	3.73×10^3	3.73×10^2	3.73×10^4	Triangular
²³² Th ^a	5.37×10^1	5.37×10^0	5.37×10^2	Triangular
²³³ U	3.82×10^3	3.82×10^2	3.82×10^4	Triangular
²³⁸ U	3.82×10^3	3.82×10^2	3.82×10^4	Triangular
²³⁸ U ^a	5.49×10^1	5.49×10^0	5.49×10^2	Triangular
²³⁹ Pu	3.90×10^3	3.90×10^2	3.90×10^4	Triangular
²⁴¹ Am	3.93×10^3	3.93×10^2	3.93×10^4	Triangular

^aValues used for Quadrex trench calculations.

Table I.6 Uncertainty values for diffusion coefficients used in the SOURCE1 and SOURCE2 computer programs

Nuclide	Diffusion coefficient of nuclide in concrete (m ² /s)			Distribution
	Probable	Low	High	
³ H	4.73×10^{-10}	9.46×10^{-11}	9.46×10^{-10}	Triangular
¹⁴ C	1.73×10^{-12}	3.46×10^{-13}	3.46×10^{-12}	Triangular
⁹⁰ Sr	1.29×10^{-13}	2.58×10^{-14}	2.58×10^{-13}	Triangular
⁹⁹ Tc	2.75×10^{-13}	5.50×10^{-14}	5.50×10^{-13}	Triangular
¹³⁷ Cs	5.12×10^{-13}	1.02×10^{-13}	1.02×10^{-12}	Triangular
²³² Th	5.04×10^{-17}	1.01×10^{-17}	1.01×10^{-16}	Triangular
²³² Th ^a	3.25×10^{-14}	6.50×10^{-15}	6.50×10^{-14}	Triangular
²³³ U	5.63×10^{-17}	1.13×10^{-17}	1.13×10^{-16}	Triangular
²³⁸ U	5.63×10^{-17}	1.13×10^{-17}	1.13×10^{-16}	Triangular
²³⁸ U ^a	3.64×10^{-14}	7.28×10^{-15}	7.28×10^{-14}	Triangular
²³⁹ Pu	6.00×10^{-17}	1.20×10^{-17}	1.20×10^{-16}	Triangular
²⁴¹ Am	4.08×10^{-17}	8.16×10^{-18}	8.16×10^{-17}	Triangular

^aValues used for Quadrex trench calculations.

**Table I.7 Uncertainty values for solubilities used in the SOURCE1
and SOURCE2 computer programs**

Nuclide	Solubility of nuclide (mol/L)			Distribution
	Probable	Low	High	
³ H	1.11×10^2	<i>a</i>	<i>a</i>	Constant
¹⁴ C	1.11×10^{-4}	<i>a</i>	<i>a</i>	Constant
⁹⁰ Sr	7.45×10^{-5}	<i>a</i>	<i>a</i>	Constant
⁹⁹ Tc	9.12×10^{-2}	<i>a</i>	<i>a</i>	Constant
¹³⁷ Cs	1.60×10^1	<i>a</i>	<i>a</i>	Constant
²³² Th	3.00×10^{-9}	3.00×10^{-10}	3.00×10^{-8}	Triangular
²³² Th ^b	7.57×10^{-8}	7.57×10^{-9}	7.57×10^{-7}	Triangular
²³³ U	3.00×10^{-10}	3.00×10^{-11}	3.00×10^{-9}	Triangular
²³⁸ U	3.00×10^{-10}	3.00×10^{-11}	3.00×10^{-9}	Triangular
²³⁸ U ^b	1.46×10^{-6}	1.46×10^{-6}	1.46×10^{-5}	Triangular
²³⁹ Pu	1.46×10^{-10}	1.00×10^{-11}	1.00×10^{-9}	Triangular
²⁴¹ Am	4.00×10^{-10}	4.00×10^{-11}	4.00×10^{-9}	Triangular

^aNot applicable. For the inventories reported to be disposed of in SWSA 6, solubility limits would not be exceeded.

^bValues used for Quadrex trench calculations.

minimum value, and maximum value of 0.15, 0.90, and 1.80 g/cm³, respectively. The mean, standard deviation, minimum value, and maximum value for the four other variables are given in Table I.8 through Table I.11. In most cases the standard deviation is 20% of the mean, and the range of values used in LHC is given by ± 3 standard deviations. The frequency distributions of values are normal in all cases except for the lognormal distributions used for the K_d s of ³H, ¹⁴C, and ⁹⁹Tc (Table I.8). These choices provide a modest range of variability for inclusion in uncertainty analyses. The five input frequency distributions are divided into 50 equal probability classes. These are sampled without replacement to generate 50 input data sets which give 50 sets of output values from the 50 simulations.

The outputs of vertical and lateral chemical transport obtained from uncertainty calculations with the SOURCE models are also needed as inputs for uncertainty analyses with the PADSIM and HOLSIM codes. These latter codes provide the uncertainty of chemical transport to groundwater used in the MOC modeling, and they also provide the shallow subsurface component of nuclide transport to surface water. These surface water results are later combined with the surface water outflow from groundwater modeling to evaluate the probability of nuclide concentrations in surface water exceeding the 4 mrem drinking water limit.

Table I.8. Attributes of the frequency distributions selected for chemical adsorption (K_d) that were used in Latin hypercube sampling

Nuclide	K_d (mL/g)	Standard deviation	Distribution	Maximum	Minimum
^3H	0.0325	0.1004	Lognormal	1	0.0001
^{14}C	0.0325	0.1004	Lognormal	1	0.0001
^{90}Sr	30	6	Normal	48	12
^{99}Tc	0.0325	0.1004	Lognormal	1	0.0001
^{137}Cs	3000	600	Normal	4800	1200
^{232}Th	3000	600	Normal	4800	1200
^{233}U	40	8	Normal	64	16
^{238}U	40	8	Normal	64	16
^{239}Pu	40	8	Normal	64	16
^{241}Am	40	8	Normal	64	16

Table I.9. Attributes of the frequency distributions selected for disposal unit area that were used in Latin hypercube sampling

Disposal unit type	Disposal unit area				
	Zone	Mean (m ²)	Standard deviation	Maximum	Minimum
High-range silos	3	10	0.50	11.50	8.50
High-range wells	5	10	0.50	11.50	8.50
High-range wells in silos	5	10	0.50	11.50	8.50
Fissile well	7	1	0.05	1.15	0.85
Low-range silos	8	10	0.50	11.50	8.50
Quadrex trenches	8	9	0.45	10.35	7.65
Biological trenches	15	47	2.40	54.20	39.80
Asbestos silos	16	10	0.50	11.50	8.50
Biological trenches	20	47	2.40	54.20	39.80
Interim Waste Management Facility	25	499	25.00	574.00	424.00
Tumulus II	26	499	25.00	574.00	424.00
Tumulus I	27	634	32.00	730.00	538.00
Low-range silos	30	10	0.50	11.50	8.50

Table I.10. Attributes of the frequency distributions selected for distance to groundwater that were used in Latin hypercube sampling

Disposal unit type	Zone	Distance to water table			
		Mean (m)	Standard deviation	Maximum	Minimum
High-range silos	3	2.9	0.58	4.64	1.16
High-range wells	5	4.3	0.86	6.88	1.72
High-range wells-in-silos	5	4.7	0.94	7.52	1.88
Fissile well	7	4.0	0.80	6.40	1.60
Low-range silos	8	3.1	0.62	4.96	1.24
Quadrex trenches	8	4.7	0.94	7.52	1.88
Biological trenches	15	0.8	0.16	1.28	0.32
Asbestos silos	16	0.6	0.12	0.96	0.24
Biological trenches	20	0.6	0.12	0.96	0.24
Interim Waste Management Facility	25	1.2	0.24	1.92	0.48
Tumulus II	26	0.5	0.10	0.80	0.20
Tumulus I	27	1.8	0.36	2.88	0.72
Low-range silos	30	2.5	0.50	4.00	1.00

Table I.11. Attributes of the frequency distributions selected for distance to a stream channel that were used in Latin hypercube sampling

Disposal unit type	Zone	Distance to stream			
		Mean (m)	Standard deviation	Maximum	Minimum
High-range silos	3	170	34.0	272.0	68.0
High-range wells	5	189	37.8	302.4	75.6
High-range wells-in-silos	5	189	37.8	302.4	75.6
Fissile well	7	142	28.4	227.2	56.8
Low-range silos	8	99	19.8	158.4	39.6
Quadrex trenches	8	99	19.8	158.4	39.6
Biological trenches	15	109	21.8	174.4	43.6
Asbestos silos	16	156	31.2	249.6	62.4
Biological trenches	20	165	33.0	264.0	66.0
Interim Waste Management Facility	25	96	19.2	153.6	38.4
Tumulus II	26	47	9.4	75.2	18.8
Tumulus I	27	47	9.4	75.2	18.8
Low-range silos	30	156	31.2	249.6	62.4

I.6 UNCERTAINTIES ASSOCIATED WITH GROUNDWATER FLOW AND CONTAMINANT TRANSPORT MODEL

The effects of parametric uncertainty on the calculation of flow and radioactive contaminant transport in groundwater were evaluated in a manner consistent with the methods used to quantify the uncertainties associated with the calculation of contaminant leaching from disposal units and their transport through the shallow subsurface. Major reasons for the parametric uncertainty in these calculations include the nature of the contaminant source (i.e., the 50 calculated contaminant flux vs time profiles for each disposal unit described in Sect. I.5) and the following nine model input variables used by the MOC code:

- anisotropy ratio ($K_{xx} : K_{yy}$),
- annual recharge,
- aquifer thickness,
- chemical adsorption coefficient (K_d),
- hydraulic conductivity,
- longitudinal dispersivity,
- ratio of transverse to longitudinal dispersivity,
- regolith bulk density, and
- regolith porosity.

Based on information in Lee (1989), which reported conductivity anisotropy ratios of ≤ 30 , the relation between strike-parallel(K_{xx}), and strike-perpendicular transmissivity (K_{yy}) were assumed to be discretely distributed in ratios of 1:1; 3:1, and 10:1. Annual recharge values were assumed to be normally distributed about the mean values reported in Sects. 4.3.1.1.2 and 4.3.1.1.4, with a standard deviation of 0.2. Aquifer thickness was assumed to be uniformly distributed with values between 3.05 and 15.24 ft (Solomon, et al. 1992). Distributions for K_d values were nuclide-specific and identical to those given in Table I.8. The logs of measured hydraulic conductivity values at SWSA 6 were analyzed and found to be normally distributed with a mean of -5.45 ft/s and standard deviation of 0.68. Based on information in Lee (1989) and on the work of Tauxe (1994) (for an average groundwater travel distance of 100 m), the log of longitudinal dispersivity was assumed to be normally distributed with a mean of 0.47 ft and a standard deviation of 1.0. The log of dispersivity anisotropy was also assumed to be normally distributed about the mean value of 1.0 (Lee 1989), with a standard deviation of 0.5 (Tauxe 1994). Consistent with Sect. I.5, regolith bulk density was assumed to be normally distributed, with a mean of 1.35 g/cm³ and a standard deviation of 0.15. Porosity was assumed to be uniformly distributed with values between 0.02 and 0.10 (Lee 1989).

Using the distributions for these nine parameters, LHC sampling was performed using the PRISM (Gardner, et al. 1983) computer code to construct 50 random input data sets per nuclide. Each of these 50 data sets, in conjunction with the corresponding 50 contaminant flux vs time profiles for each disposal unit (described in Sect. I.5), were then used as input for nuclide-specific

simulations performed using the MOC model. Unlike the previous uncertainty analyses in which separate simulations were performed for each disposal unit, all units associated with a given nuclide were analyzed simultaneously in each of the 50 MOC runs. The results of this analysis yielded distributions of 50 values of maximum groundwater concentration (for each radionuclide) at each of the 179 groundwater receptor locations for each radionuclide. Furthermore, 50 distributions of contaminant flux to surface water due to contaminant transport in groundwater were also generated. These were combined with the 50 distributions of contaminant flux to surface water due to shallow subsurface transport (Sect. I.5) to obtain distributions of total contaminant concentrations in surface water.

I.7 SENSITIVITY/UNCERTAINTY ANALYSIS FOR DOSE ASSESSMENT MODELS

This section discusses the sensitivity and uncertainty analysis for the exposure pathway and dose assessment models used in estimating annual effective dose equivalents (EDEs) to off-site individuals from releases of radionuclides in groundwater and surface water. This discussion is based on the models and results presented in Appendix G, Sect. G.5.1. The sensitivity and uncertainty analysis for the models used to estimate doses to inadvertent intruders for the various disposal units in the SWSA 6 facility is discussed separately in Sect. 6.4.

I.7.1 Discussion of Dose Assessment Models

In the dose assessment for releases of radionuclides in water, the assumed exposure pathways for off-site individuals include direct ingestion of contaminated water, ingestion of milk and meat obtained from dairy and beef cattle that drink contaminated water, ingestion of fish obtained from contaminated water, and external exposure while swimming in contaminated water. The drinking water, milk, and meat pathways apply to contaminated groundwater and surface water, but the fish and swimming pathways apply only to contaminated surface water.

Simple multiplicative-chain models are used to estimate doses to off-site individuals per unit concentration of radionuclides in water at assumed receptor locations. The doses per unit concentration of radionuclides for the different exposure pathways are used in conjunction with the performance objectives for limiting dose to off-site individuals from the drinking water pathway only and from all exposure pathways combined to derive maximum allowable concentrations of radionuclides in groundwater and surface water at off-site locations.

In the model for any exposure pathway, the estimated dose per unit concentration of a radionuclide in water is a linear function of the input parameters, which include transfer factors for radionuclides through food chain pathways (milk, meat, and fish pathways), intakes of water and foodstuffs (drinking water, milk, meat, and fish pathways) or the exposure time (swimming pathway), and doses per unit intake of radionuclides by ingestion (drinking water, milk, meat, and

fish pathways) or dose rates per unit concentration of radionuclides in water (swimming pathway). The sensitivity analysis for a linear model is particularly simple because a change in a particular parameter value by an assumed factor, with all other parameter values remaining fixed, results in a change in the model output by the same factor.

I.7.1.1 Uncertainty in Dose Assessment Models for Groundwater Pathways

As described above, only the drinking water, milk, and meat pathways are assumed to result from exposure to contaminated groundwater. The doses per unit concentration of radionuclides in water for these pathways are given in Appendix G, Table G.7, Table G.10, and Table G.11.

For nearly all radionuclides considered in this assessment, the dose per unit concentration in water is considerably higher for the drinking water pathway than for the milk and meat pathways, and the dose from all exposure pathways from use of contaminated groundwater thus is determined almost entirely by the dose from the drinking water pathway. In the worst case (i.e., for ^{94}Nb), the dose from the milk and meat pathways is almost a factor of two higher than the dose from the drinking water pathway. Thus, in the unusual cases where the milk and meat pathways are important, the dose from all exposure pathways exceeds the dose from the drinking water pathway by less than a factor of three for the parameter values assumed in this assessment.

Furthermore, different performance objectives are applied to exposures of off-site individuals from the drinking water pathway and from all exposure pathways. Specifically, the performance objective for the drinking water pathway of 4 mrem/year EDE is about six times more restrictive than the performance objective for all exposure pathways of 25 mrem/year EDE.

Therefore, taking into account the doses per unit concentration of radionuclides in water for the different exposure pathways and the difference in the performance objectives for the drinking water pathway and all exposure pathways, the maximum allowable concentrations of all radionuclides in groundwater at off-site locations are determined entirely by the dose per unit concentration for the drinking water pathway only, given the parameter values assumed in the dose assessment models.

The dose from the drinking water pathway per unit concentration of a radionuclide in water depends on two parameters: the assumed intake of contaminated water and the ingestion dose conversion factor for the radionuclide [see Appendix G, Eq. (G.1)]. In any population of exposed individuals, these parameters would be variable quantities and, thus, would be subject to uncertainty for any real individual. However, for purposes of performance assessment or, more generally, any type of prospective dose assessment involving hypothetical exposures of reference individuals, both of these parameters normally are assumed to be fixed quantities, without uncertainty, that are prescribed by regulatory authorities and radiation protection experts. That is, consistent with current practices in radiation protection and in applying drinking water standards, dose assessments for the drinking water pathway are based on prescribed intakes of water and prescribed ingestion dose conversion factors for radionuclides, and no uncertainty is assigned to these parameters. Therefore,

for purposes of performance assessment, the dose from the drinking water pathway per unit concentration of radionuclides in water essentially has no uncertainty.

As discussed previously, the dose from the milk and meat pathways is unimportant compared with the dose from the drinking water pathway for all radionuclides, given the parameter values assumed in this assessment. However, the dose from the milk and meat pathways is subject to uncertainty because of uncertainties in the intake-to-milk and intake-to-meat transfer coefficients of radionuclides for dairy and beef cattle [see Appendix G, Eq. (G.2) through Eq. (G.5)]. All other parameters in the models for these pathways are fixed quantities prescribed by regulatory authorities and radiation protection experts and, thus, can be assumed to have no uncertainty. If the transfer coefficients for the milk and meat pathways were considerably higher than the values assumed in this assessment, the dose from all exposure pathways could be more important than the dose from the drinking water pathway and the uncertainty in the dose from all exposure pathways could result in an uncertainty in the maximum allowable concentrations of radionuclides in groundwater derived from the dose assessment.

As noted previously, the milk and meat pathways are important compared with the drinking water pathway only for a very few radionuclides. Data on the transfer coefficients for the milk and meat pathways for these radionuclides are rather sparse [see Baes et al. (1984) and references therein] and, thus, it is difficult to estimate uncertainties in the assumed values. For this discussion, it is assumed that the transfer coefficients for ^{94}Nb , which is the radionuclide with the highest dose per unit concentration in water for the milk and meat pathways, could be an order of magnitude higher than the values used in the dose assessment. This assumption, although somewhat arbitrary, may be reasonable for radionuclides for which few measurements are available because uncertainties in the transfer coefficients for well-studied radionuclides (e.g., ^{90}Sr , ^{131}I , and ^{137}Cs) appear to be substantially less than an order of magnitude [see NRC and CEC (1997)]. An increase in the transfer coefficients for ^{94}Nb by a factor of 10 would result in a dose from all exposure pathways that is a factor of about 20 higher than the dose from the drinking water pathway. Taking into account the difference of a factor of 6 between the performance objectives for the drinking water pathway and all exposure pathways, the maximum allowable concentration of ^{94}Nb in water then would be determined by the dose per unit concentration from all exposure pathways, and the maximum allowable concentration would be about a factor of 3 less than the value determined by the dose per unit concentration from the drinking water pathway alone.

Therefore, in the worst case, taking into account uncertainties in the transfer coefficients of radionuclides for the milk and meat pathways probably would affect the maximum allowable concentrations of radionuclides in groundwater by no more than a factor of about 3. Such an uncertainty clearly is insignificant compared with uncertainties in the models for estimating the maximum concentrations of radionuclides in groundwater at off-site locations (i.e., uncertainties in estimated releases from disposal units and transport in the environment) and, in the case of the closed disposal units, uncertainties in the inventories of radionuclides disposed. Furthermore, for most radionuclides, including those that are expected to be the most important in wastes disposed in SWSA 6, the dose from the milk and meat pathways is sufficiently less than the dose from the

drinking water pathway that uncertainties in the transfer coefficients for the milk and meat pathways should not affect the maximum allowable concentrations in groundwater derived from the dose assessment, again because the performance objective for the drinking water pathway is considerably more restrictive than the performance objective for all exposure pathways.

I.7.1.2 Dose Assessment Models for Surface Water Pathways

As described previously, the drinking water, milk, meat, fish, and swimming pathways are assumed to result from exposure to contaminated surface water. The doses per unit concentration of radionuclides in water for these pathways are given in Appendix G, Table G.7, Table G.10, Table G.11, Table G.13, and Table G.14. In addition, the fish pathway is assumed to be credible only for exposure to water in the Clinch River, but the other exposure pathways are applied in White Oak Creek, where the concentrations of radionuclides are assumed to be a factor of 330 higher than the concentrations in the Clinch River. Thus, off-site individuals receiving the highest doses are assumed to obtain fish from the Clinch River but are assumed to be exposed to water in White Oak Creek for the other pathways.

The discussions in the previous section about the importance of the milk and meat pathways relative to the drinking water pathway for radionuclides in groundwater are directly applicable to exposures to contaminated surface water. Thus, based on the assumed exposure pathway models and parameters and taking into account the difference between the performance objectives for the drinking water pathway and all exposure pathways, the milk and meat pathways are unimportant relative to the drinking water pathway in determining the maximum allowable concentrations of all radionuclides in surface water, and the maximum allowable concentrations derived from the dose assessment for the drinking water pathway essentially have no uncertainty. Furthermore, if uncertainties in the transfer coefficients in the models for the milk and meat pathways are taken into account, the maximum allowable concentrations in surface water could be reduced by a factor of about 3 in the worst case, but consideration of these uncertainties would not significantly change the maximum allowable concentrations for most radionuclides.

For all radionuclides, the dose from the swimming pathway per unit concentration in water is much less than the dose from the drinking water pathway. In the model for the swimming pathway [see Appendix G, Eq. (G.8)], the only uncertain parameter is the exposure time. However, even if the actual exposure time were an order of magnitude higher than the value of 100 h/year assumed in this assessment, which would be unreasonable, the dose from the swimming pathway still would be substantially less than the dose from the drinking water pathway for all radionuclides. Therefore, given the difference between the performance objectives for the drinking water pathway and all exposure pathways, the uncertainty in the dose from the swimming pathway can have no effect on the maximum allowable concentrations of radionuclides in surface water derived from the dose assessment.

For several radionuclides, the dose from the fish pathway per unit concentration in water is considerably higher than the dose from the drinking water pathway. For ^{14}C , which is the worst case,

the difference is a factor of 35. However, as noted previously, an off-site individual is assumed to obtain drinking water from White Oak Creek but to consume fish from the Clinch River, and the concentrations of radionuclides in the river are assumed to be a factor of 330 less than the concentrations in the creek. Therefore, for the parameter values assumed in this assessment, the dose from the fish pathway would be at least an order of magnitude less than the dose from the drinking water pathway for all radionuclides and, thus, would be insignificant.

The only uncertain parameter in the model for the fish pathway is the transfer coefficient from water to fish [see Appendix G, Eq. (G.6) and Eq. (G.7)]. Uncertainties in this transfer coefficient for different radionuclides are unknown. However, even if the transfer coefficients were an order of magnitude higher than the values assumed in this assessment, the dose from the fish pathway would be no higher than the dose from the drinking water pathway in the worst case, and would be considerably less for most radionuclides. Therefore, given the difference between the performance objectives for the drinking water pathway and all exposure pathways, it is highly unlikely that uncertainties in the transfer coefficients of radionuclides for the fish pathway would have any effect on the maximum allowable concentrations of radionuclides in surface water derived from the dose assessment.

I.7.1.3 Summary of Uncertainties in Dose Assessment Models

Based on the discussions in the previous two sections, it is evident that uncertainties in the dose assessment models for exposure of off-site individuals to contaminated groundwater or surface water are insignificant for this PA, especially in comparison with uncertainties in radionuclide inventories in disposal units and uncertainties in the models for estimating release of radionuclides from disposal units and transport to assumed off-site receptor locations. The key factors leading to this conclusion are summarized as follows.

First, for the models and parameter values assumed in this assessment, the dose from the drinking water pathway is higher than the dose from all other exposure pathways combined for most of the radionuclides considered in this assessment. For those few radionuclides for which the dose from the other exposure pathways is unusually high, the dose from all exposure pathways exceeds the dose from the drinking water pathway by only about a factor of 3.

Second, the performance objective for the drinking water pathway for off-site individuals is about a factor of 6 more restrictive than the performance objective for all exposure pathways. Thus, for all radionuclides, the difference between the two performance objectives is greater than the difference between the dose from all exposure pathways and the dose from the drinking water pathway only, and the maximum allowable concentrations of radionuclides in groundwater and surface water can be based entirely on the dose assessment for the drinking water pathway, which essentially has no uncertainty.

Finally, even if the dose from the exposure pathways other than drinking water were an order of magnitude higher than assumed in this assessment, based on consideration of uncertainties in the transfer coefficients for the milk, meat, and fish pathways, the resulting maximum allowable

concentrations of radionuclides in water would be reduced by no more than a factor of 3 in the worst case and by considerably less for most radionuclides. The magnitude of any such uncertainties clearly is insignificant for this PA.

I.7.2 General Considerations of Uncertainties in Dose Assessment Models

In evaluating the importance of the different sources of uncertainty in the dose assessment models for off-site individuals (Sect. I.7.1), it also should be recognized that an important source of uncertainty is the definitions of the exposure scenarios themselves, notwithstanding any parameter uncertainties that could be quantified and regardless of whether the results would represent the variability in doses that could be received by individuals near the disposal site. The dose assessments for exposure of off-site individuals to contaminated groundwater or surface water are based on assumptions that the scenarios will occur as postulated, but many of the assumptions used in defining and implementing the scenarios are conservative.

Perhaps the most important assumption that is likely to be conservative is the assumption that off-site individuals will be exposed at those locations where the highest concentrations of radionuclides in water are calculated to occur. This is particularly the case for exposure to contaminated groundwater. In addition, the assumptions about the quantities of contaminated drinking water, milk, meat, and fish that would be consumed by individuals should overestimate the exposures for these pathways that would be experienced by most individuals [e.g., see Rupp (1980) and Yang and Nelson (1986)]. Thus, the exposure scenarios for off-site individuals assume conditions that tend to overestimate the doses that would be received by most individuals.

More fundamentally, it is not the purpose of a dose assessment for off-site individuals to provide best estimates of doses that likely would be received and to quantify the uncertainties in these estimates. Rather, the primary purpose of the exposure pathway modeling is to determine the maximum allowable concentrations of radionuclides in groundwater or surface water that would be adequately protective of off-site individuals, without undue concern for the magnitude of doses that any individuals might actually receive. Furthermore, quantitative estimates of the uncertainty in calculated doses per unit concentration of radionuclides in water based on parameter uncertainty analysis may not be particularly meaningful because the results are conditional on the occurrence of the assumed exposure scenarios at the locations of maximum radionuclide concentrations in water. Therefore, the most important factor in determining whether the maximum allowable concentrations in water derived from the dose assessment are reasonable is the credibility of the assumed exposure scenarios and pathways (i.e., whether the assumed scenarios and pathways reasonably could occur near the disposal site) rather than any estimates of uncertainties in the results due to uncertainties in the parameters used in the dose assessment models.

I.8 RESULTS

I.8.1 Results of Parametric Uncertainty Analysis

Results that illustrate the effects of parametric uncertainties on predicted radionuclide concentrations in groundwater and surface water are presented in the following section. Fig. I.2 through Fig. I.11 depict histograms and the associated CDFs for the maximum groundwater concentration (at the receptor location corresponding to the maximum groundwater concentration calculated during the deterministic analysis) obtained from each of the 50 LHC iterations. Similarly, Fig. I.12 through Fig. I.21 show histograms and CDFs for the maximum surface water concentration at White Oak Dam obtained from each of the same 50 LHC model iterations. For reference purposes, the maximum concentration (groundwater or surface water) calculated from the deterministic analysis and the drinking water concentration corresponding to the 4 mrem/year dose limit (C^{lim}) are included in both sets of figures.

Measures of uncertainty associated with these results include the spread of the PDF or the associated entropy-based value (determined by Eq. I.1). Although the PDFs displayed in Fig. I.2 through Fig. I.21 represent values determined using inventories for the groundwater limiting scenario, similar results would be expected had inventories for other surface-water-limiting cases been used. Based on the distributions of maximum groundwater concentrations (Fig. I.2 through Fig. I.11), for many radionuclides, the use of the disposal limits from the deterministic analysis presented in Sect. 5 provides reasonable assurance that the performance objectives of DOE O 5820.2A will be met. This is true for radionuclides such as ^3H with extremely small K_d values or for those with relatively long half-lives and moderate K_d values such as ^{233}U . However, for radionuclides with combinations of large K_d values and short half-lives (^{137}Cs), or those with extremely large K_d values and half-lives (^{232}Th), the results suggest that considerable uncertainty exists in the calculated values found in Sect. 5.1.2.2. Hence, simply basing disposal limits on the results of the deterministic analysis is not sufficient to ensure compliance with performance objectives. Instead, the parametric uncertainty analysis suggests that it is necessary to modify the limits presented in Sect. 5.1.2.2, based on the results of this uncertainty analysis, to obtain "reduced" disposal limits that do provide a reasonable probability that SWSA 6 is in compliance.

I.8.2 Results of Subjective Uncertainty Analysis

The following section summarizes the results from the subjective uncertainty analysis in which probability distributions obtained from the parametric uncertainty analysis (Sect. I.8) were combined with the subjective estimates presented in Sect. I.1.3.2.3. The effects of subjective uncertainties are illustrated by showing their impact on the probability that predicted environmental radionuclide concentrations are less than some reference (in this case, the 4 mrem/year drinking water limit, C^{lim}) value. Table I.12 lists the probabilities and the associated entropy-based uncertainties, based on results of the parametric uncertainty analysis, that groundwater

concentrations for the five radionuclides evaluated in the subjective uncertainty analysis are less than C^{lim} , as well as the modified values due to the inclusion of subjective uncertainties. Similarly, Table I.13 lists the "before and after" probabilities and entropy-based uncertainties that total surface water concentrations for these radionuclides are less than C^{lim} . In addition, Table I.14 and Table I.15 provide a more detailed account of the effects of subjective uncertainties. These tables show the changes in the probability that the deterministic result is within the drinking water performance objective as the subjective uncertainties associated with each phase of the modeling process are incorporated.

First, it is of interest to note the differences between the initial probabilities that the deterministic result is within the drinking water performance objective given in Table I.12 and Table I.13 and the corresponding values given in Table 4.35 and Table 4.36 of the SWSA 6 1994 PA (ORNL 1994). With the exception of ^{14}C , the inventories analyzed in the current analysis were significantly higher than those analyzed in the 1994 analysis. Consequently, with the exception of ^3H (in groundwater) and ^{90}Sr (in surface water), the initial probabilities that the deterministic result is within the drinking water performance objective determined from the parametric analysis are now equal to or lower than those calculated in the 1994 analysis. In the case of ^3H , probabilities presented in the 1994 analysis are based on the maximum groundwater concentration that occurs at any point in time. For a given inventory, these concentrations are significantly higher than the maximum concentration that occurs after the 100-year institutional control period (as were used in the current analysis). For ^{90}Sr , differences between results calculated during the current analysis and those presented in the 1994 analysis are attributed to changes in the PADSIM/HOLSIM models. For ^{14}C , the inventory analyzed in the current analysis is less than that used in the 1994 analysis. Hence the initial probabilities that the deterministic result is within the drinking water performance objective increased in both groundwater and surface water.

Considering the effects of parametric uncertainties, Sect. I.8 illustrated that, for some radionuclides, basing disposal limits on values calculated using the CTM may not be sufficient to ensure SWSA 6 complies with the performance objectives of DOE O 5820.2A. The inclusion of subjective uncertainties further illustrates the limitations of any modeling procedure, such as that used in this PA, in accurately predicting environmental concentrations of radioactive contaminants. For example, Table I.12 and Table I.13 indicate how the convolution of subjective opinion with parametric results from the LHC analysis tends to drive the probability that radionuclide concentrations in groundwater and surface water are less than C^{lim} toward 0.5 for all radionuclides. Hence, the associated uncertainty [based on Eq. (I.1)] approaches the maximum value of 1.0. However, the effect of time on the magnitude of the uncertainty increase associated with model predictions is demonstrated by the changes in uncertainty shown in the last columns of Table I.12 and Table I.13. Generally, the radionuclides that experience the greatest increase in uncertainty with the inclusion of subjective opinion are those whose peak concentrations occur at large time values. For example, ^3H groundwater and surface water concentrations reach maximum values within about 100 years from the time of disposal. And, as expected, there is only a relatively small increase in the associated uncertainty in model predictions. Conversely, environmental concentrations of ^{238}U ,

which experience an increase in uncertainty of 0.94, reach their maximum values thousands of years after the time of disposal. Thus, inclusion of subjective uncertainty effects emphasizes the fact that accurate determination of disposal limits based on environmental transport of radioactive contaminants is a difficult, subjective process that becomes even more complicated as the environmental travel time of contaminants increases.

Subjective uncertainty may also be used to identify the segments of modeling processes that are the chief contributors to the total uncertainty in predicted values, thereby identifying areas that would benefit most from further research. For example, consider the change in the probabilities that the deterministic result is within the drinking water performance objective (and, therefore, the change in overall uncertainty) that occurs with the successive convolutions of the subjective uncertainties associated with each component of the model process with the concentration distributions obtained from the parametric uncertainty analysis. The results presented in Table I.14 (groundwater) and Table I.15 (surface water) show that, in all instances, inclusion of the subjective uncertainties associated with disposal inventory results in the greatest overall increase in the uncertainty associated with predicted radionuclide concentrations. Therefore, the logical first step to reduce the overall uncertainty associated with predicted radionuclide concentrations would be to develop a more reliable method for estimating the inventory of disposed radioactive material.

³H Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Groundwater Concentrations at Observation Well 16

Fig. I.2. ³H uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

^{14}C Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Groundwater Concentrations at Observation Well 16

Fig. I.3. ^{14}C uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

**^{90}Sr Uncertainty Analysis: Frequency and Cumulative Distributions of
Maximum Groundwater Concentrations
at Observation Well 17**

Fig. I.4. ^{90}Sr uncertainty analysis: Histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 17.

**^{99}Tc Uncertainty Analysis: Frequency and Cumulative Distributions of
Maximum Groundwater Concentrations
at Observation Well 16**

Fig. I.5. ^{99}Tc uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

**^{137}Cs Uncertainty Analysis: Frequency and Cumulative Distributions of
Maximum Groundwater Concentrations
at Observation Well 52**

Fig. I.6. ^{137}Cs uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 52.

^{232}Th Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Groundwater Concentrations at Observation Well 16

Fig. I.7. ^{232}Th uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

^{233}U Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Groundwater Concentrations at Observation Well 16

Fig. I.8. ^{233}U uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

^{238}U Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Groundwater Concentrations at Observation Well 16

Fig. I.9. ^{238}U uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

**^{239}Pu Uncertainty Analysis: Frequency and Cumulative Distributions of
Maximum Groundwater Concentrations
at Observation Well 16**

Fig. I.10. ^{239}Pu uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

²⁴¹Am Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Groundwater Concentrations at Observation Well 16

Fig. I.11. ²⁴¹Am uncertainty analysis: histogram and cumulative distribution function of maximum groundwater concentration at Observation Well 16.

³H Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.12. ³H uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

¹⁴C Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.13. ¹⁴C uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

⁹⁰Sr Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.14. ⁹⁰Sr uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

^{99}Tc Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.15. ^{99}Tc uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

¹³⁷Cs Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.16. ¹³⁷Cs uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

²³²Th Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.17. ²³²Th uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

²³³U Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.18. ²³³U uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

²³⁸U Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.19. ²³⁸U uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

²³⁹Pu Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.20. ²³⁹Pu uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

²⁴¹Am Uncertainty Analysis: Frequency and Cumulative Distributions of Maximum Surface Water Concentrations

Fig. I.21. ²⁴¹Am uncertainty analysis: histogram and cumulative distribution function of maximum surface water concentration at White Oak Dam.

Table I.12. Probabilities and associated entropy-based uncertainties that maximum groundwater concentrations at selected receptor locations $\leq$ 4 mrem drinking water limit

Nuclide	Receptor ID	4 mrem drinking water limit ($\mu\text{Ci/L}$)	Parametric		Parametric + Subjective		ΔU
			P_c	U	P_c	U	
^3H	16	8.50×10^{-2}	0.90	0.47	0.60	0.97	+0.50
^{14}C	16	2.70×10^{-3}	0.90	0.47	0.53	0.99	+0.52
^{90}Sr	17	3.60×10^{-5}	0.38	0.96	0.46	0.99	+0.03
^{137}Cs	52	1.10×10^{-4}	1.00	0.00	0.82	0.69	+0.69
^{238}U	16	2.10×10^{-5}	1.00	0.00	0.64	0.94	+0.94

Table I.13. Probabilities and associated entropy-based uncertainties that maximum surface water concentrations at selected receptor locations $\leq$ 4 mrem drinking water limit

Nuclide	4 mrem drinking water limit ($\mu\text{Ci/L}$)	Parametric		Parametric + Subjective		ΔU
		P_c	U	P_c	U	
^3H	8.50×10^{-2}	0.92	0.40	0.45	0.99	+0.55
^{14}C	2.70×10^{-3}	1.00	0.00	0.32	0.91	+0.91
^{90}Sr	3.60×10^{-5}	0.94	0.33	0.50	1.00	+0.77
^{137}Cs	1.10×10^{-4}	1.00	0.00	0.65	0.94	+0.94
^{238}U	2.10×10^{-5}	1.00	0.00	0.42	0.98	+0.98

Table I.14. Changes in the probabilities and associated entropy-based uncertainties that maximize groundwater concentrations at selected receptor locations ≤ 4 mrem drinking water limit due to inclusion of subjective uncertainties associated with each segment of the modeling process

Nuclide	(1)		(2)		(3)		(4)		(5)		(6)			
	Initial Values from Parametric Uncertainty Analysis		(1) + Subjective Uncertainties Associated with Inventory		(2) + Subjective Uncertainties Associated with UTM		(3) + Subjective Uncertainties Associated with Source1/Source2		(4) + Subjective Uncertainties Associated with PADSIM/HOLSIM		(5) + Subjective Uncertainties Associated with MOC			
	P _c	U	P	c	U	P	c	U	P	c	U	P	c	U
³ H	0.90	0.47	0.69		0.89	0.67		0.91	0.65		0.93	0.61		0.97
¹⁴ C	0.90	0.47	0.57		0.98	0.56		0.99	0.55		0.99	0.54		0.99
⁹⁰ Sr	0.38	0.96	0.44		0.99	0.44		0.99	0.45		0.99	0.45		0.99
¹³⁷ Cs	1.00	0.00	0.90		0.47	0.86		0.59	0.86		0.59	0.82		0.69
²³⁸ U	1.00	0.00	0.70		0.88	0.68		0.91	0.67		0.91	0.65		0.93
												0.64		0.94

Table I.15. Changes in the probabilities and associated entropy-based uncertainties that maximize surface water concentrations at selected receptor locations ≤ 4 mrem drinking water limit due to inclusion of subjective uncertainties associated with each segment of the modeling process

Nuclide	(1)		(2)			(3)			(4)			(5)			(6)		
	Initial Values from Parametric Uncertainty Analysis		(1) + Subjective Uncertainties Associated with Inventory			(2) + Subjective Uncertainties Associated with UTM			(3) + Subjective Uncertainties Associated with Source1/Source2			(4) + Subjective Uncertainties Associated with PADSIM/HOLSIM			(5) + Subjective Uncertainties Associated with MOC		
	P _c	U	P	c	U	P	c	U	P	c	U	P	c	U	P	c	U
³ H	0.92	0.40	0.53		0.99	0.49		0.99	0.49		0.99	0.45		0.99	0.45		0.99
¹⁴ C	1.00	0.00	0.35		0.94	0.34		0.93	0.34		0.92	0.32		0.91	0.32		0.91
⁹⁰ Sr	0.94	0.33	0.61		0.97	0.57		0.99	0.56		0.99	0.52		0.99	0.50		1.00
¹³⁷ Cs	1.00	0.00	0.81		0.70	0.74		0.83	0.73		0.84	0.65		0.94	0.65		0.94
²³⁸ U	1.00	0.00	0.48		0.99	0.46		0.99	0.45		0.99	0.43		0.98	0.42		0.98

I.9 REFERENCES

- Baes, C. F. III, et al. 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides Through Agriculture*, ORNL-5786, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Begovich, C. L., and R. J. Luxmoore. 1979. *Some Sensitivity Studies of Chemical Transport Simulated in Models of the Soil-Plant-Litter System*. ORNL/TM-6791. Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Gardner, R. H., B. Røjder, and U. Bergström 1983. *PRISM, A Systematic Method for Determining the Effect of Parameter Uncertainties on Model Predictions*, Report NW-83-555, Studsvik Energiteknik AB, Nyköping, Sweden.
- Icenhour, A. S., and M. L. Tharp 1995. "Recent Improvements to the SOURCE1 and SOURCE2 Computer Codes," in *Proceedings of the U.S. Department of Energy 17th Annual Low-Level Radioactive Waste Management Conference*, Phoenix, Arizona, December 12-14, 1995, Idaho National Engineering Laboratory, Idaho Falls, Idaho.
- Icenhour, A. S., M. L. Tharp, and L. F. Miller 1997. "Time-Dependent Sensitivity Analysis of the SOURCE1 Computer Code," *Transactions of the American Nuclear Society*, 76, 192-3.
- Lee, R. R., et al. 1991. "Aquifer Analysis and Modeling in a Fractured Heterogeneous Medium," *Ground Water* 30(4), 589-97.
- Lee, R. R., et al. 1989. *Calibration of a Groundwater Flow and Contaminant Transport Model: Progress Toward Model Validation*, ORNL/TM-11294, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Luxmoore, R. J. 1982. *Physical Characteristics of Soils of the Southern Region*, ORNL-5868. Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Luxmoore, R. J., J. L. Stolzy, and J. T. Holdeman. 1976. *Some Sensitivity Analyses of an Hourly Soil-Plant-Water Relations Model*, ORNL/TM-5343. Oak Ridge National Laboratory, Oak Ridge, Tenn.
- NRC and CEC (U.S. Nuclear Regulatory Commission and Commission of the European Communities) 1997. *Probabilistic Accident Consequence Uncertainty Analysis: Food Chain Uncertainty Assessment*, NUREG/CR-6523, EUR 16771, SAND97-0335, Vol. 1 and 2, CEC, Luxembourg.
- ORNL (Oak Ridge National Laboratory) 1994. *Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6*, ORNL-6783, Oak Ridge National Laboratory, Oak Ridge, Tenn.

- Rupp, E. M. 1980. "Age Dependent Values of Dietary Intake for Assessing Human Exposures to Environmental Pollutants," *Health Physics* **39**, 151.
- Sharma, M. L., and R. J. Luxmoore. 1979. "Soil Spatial Variability and Its Consequence on Simulated Water Balance," *Water Resources Research* **15**, 1567-1573.
- Solomon, D. K., et al. 1992. *Status Report: A Hydrologic Framework for the Oak Ridge Reservation*, ORNL/TM-12026, Oak Ridge National Laboratory, Oak Ridge, Tenn.
- Tauxe, J. D. 1994. *Porous Medium Advection-Dispersion Modeling in a Geographic Information System*, Technical Report CRWR 253, Center for Research in Water Resources, The University of Texas at Austin, Austin, Tex.
- Yang, Y.-Y., and C. B. Nelson. 1986. "An Estimation of Daily Food Usage Factors for Assessing Radionuclide Intakes in the U.S. Population," *Health Physics* **50**, 245.

Appendix J

RESPONSES TO COMMENTS AND RECOMMENDATIONS

D. W. Lee

Appendix J

RESPONSES TO COMMENTS AND RECOMMENDATIONS:

Responses to

DOE Headquarters Review of the "Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6"

and the

Performance Assessment Peer Review Panel Recommendation on the "Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6,"

ORNL-6783, February, 1994

The *Performance Assessment for Continuing and Future Operations of Solid Waste Storage Area 6* (ORNL-6783), February 1994, was submitted to DOE Headquarters for review by the Peer Review Panel on March 24, 1994. The review was concluded on January 25, 1995, with the finding that the performance assessment (PA) and the supplemental information were technically acceptable. The Peer Review Panel submitted several recommendations to DOE Headquarters for consideration. The Office of Waste Management, DOE Headquarters, completed its review of the PA on September 8, 1995, and concluded that the PA was technically adequate. Headquarters also concluded that several actions should be taken with respect to the PA and that these should be incorporated into a revision of the PA to be resubmitted to Headquarters.

In response to the Defense Nuclear Facilities Safety Board (DNFSB) Recommendation 94-2, Headquarters directed on May 31, 1995, that the PA be revised to consider additional source terms from wastes disposed of prior to September 26, 1988. This directive was amended to require the preparation of a composite analysis (CA) to address all contributing sources of contamination either in a separate document or as part of the PA, subject to guidance issued on October 31, 1996. This appendix responds to the comments made by Headquarters in the review of the PA and to the comments made by the Peer Review Panel. Consideration of contributing sources of contamination

from wastes disposed of prior to September 26, 1988, is addressed in the CA for SWSA 6, which is a separate document.

J.1 DOE HEADQUARTERS COMMENTS

The SWSA 6 PA should be revised and resubmitted for Headquarters approval. The revision should address the following items:

Item: *Impacts of actions taken as a result of the PA on continuing operation of the Interim Waste Management Facility (IWMF) have not been analyzed. Thus, the PA presently does not provide documented assurance that continuing IWMF operation will comply with performance objectives defined in DOE Order 5820.2A.*

Response: The revised PA addresses all changes made in operations following the publication of the initial version of the PA and provides concentration and inventory limits for radionuclides in wastes for continued IWMF operations. The limits provided in this revised PA were determined to ensure that continuing operations will be in compliance with the performance objectives of DOE Order 5820.2A. The limits provided in this revised PA will be incorporated in the Waste Acceptance Criteria for IWMF operations.

Item: *The Department is adopting new requirements for the scope of low-level waste disposal facility PAs in response to Defense Nuclear Facilities Safety Board (DNFSB) recommendation 94-2. The new requirements are that a PA for a low-level waste disposal facility include impacts from other disposed waste and other sources of radioactive contamination in the ground (leaks, etc.) that potentially contribute to the doses from the disposal facility in the analysis. The results of the analysis are then to be compared with the performance objectives of the Order.*

Response: The CA for SWSA 6 has been prepared according to the guidance issued on October 31, 1996, and is being submitted to Headquarters coincident with the submittal of this revised PA. The CA considers all sources of contamination in the ground that could potentially contribute to the doses from SWSA 6.

Item: *Supplemental information developed during the course of the review by the Peer Review Panel should be incorporated into the revision. Additionally, detailed comments on the PA and supplemental information developed by the Peer Review Panel should be considered as the revision is prepared.*

Response: All supplemental information developed in the review of the SWSA 6 PA has been incorporated into this revision of the PA. Additional information developed since the

publication of the initial version of the PA has also been incorporated into this revision of the PA. The written comments made by the Peer Review Panel at the conclusion of its review are addressed below. Comments made by the Peer Review Panel in its review of the PA and the supplemental information that is recorded in the minutes of the Peer Review Panel meetings have been considered in this revision of the PA.

J.2 PEER REVIEW PANEL COMMENTS

Comment: *The Panel feels strongly that there is an urgent need for revision of the PA. The revision is needed to provide assurance that continuing operation of the Interim Waste Management Facility (IWMF) will comply with DOE performance objectives.*

Response: This revision of the PA provides the needed assurance that IWMF operations will comply with DOE performance objectives by establishing radionuclide concentration and inventory limits for wastes to be disposed of at the IWMF. These limits were defined so the performance objectives would not be exceeded at the IWMF. The limits calculated in this revision of the PA will be incorporated into the Waste Acceptance Criteria for the IWMF.

Comment: *The PA revision is also needed to document the effect of waste management operational actions taken due to the results of the PA and the supplemental information developed during the review.*

Response: All of the changes in IWMF operations that have occurred since the publication of the SWSA 6 PA have been incorporated into this revision of the PA. The supplemental information developed during the review of the PA has also been incorporated into this PA. Additional information that has been developed since the conclusion of the PA reviews has also been included. Finally, the revised PA has been restructured to adhere to the guidance for the preparation of PAs issued on November 1, 1996.

Comment: *The Panel would like to commend the Oak Ridge Field Office and ORNL personnel for waste management actions taken as a result of the PA.*

Response: No response required.

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