TO: Paul H. Lohaus, STP Rec'd STP: 12/11/2003

FROM: Kristin Felix, State of Washington

(from Final EIS, Chapter 2)

2.3.1.3 High-Level Radioactive Waste

The commercial LLRW site has never been licensed to receive high-level radioactive waste. However, during the 1970's, before the federal government distinguished between high and low-level waste, approximately 13,800 curies of irradiated fuel segments and other spent fuel waste was disposed at the commercial LLRW site (DOH 2003b). Today, this waste would be classified as high-level waste. This waste was disposed under the authority of the USNRC. The USNRC also confirmed that, under their authority, there is a small chance that two fuel rods from a Connecticut Nuclear Power Plant named Millstone Unit 1 were inadvertently disposed at the commercial LLRW site in 1988 (USNRC 2003). The fuel rods are highly radioactive and are classified as high-level waste.

In January 2003, the USNRC issued a Draft Safety Evaluation Report for Millstone Unit 1 (USNRC 2003). Millstone Unit 1 had reported that the location of two fuel rods could not be determined. The safety analysis stated that the most likely disposal scenario was for the rods to have been inadvertently shipped to the commercial LLRW disposal site in Barnwell, South Carolina in 1988. However, the safety analysis also concluded that there was a small chance that the rods may have unintentionally been buried at the commercial LLRW site here in Washington.

The safety analysis concluded that if the two fuels rods were disposed at the Richland commercial LLRW site, it would not constitute a present or future risk to the public health and safety nor to the environment. This conclusion was also supported by USEPA (USEPA 2003). Assuming the fuel rods were shipped to Richland, the waste type would have been misidentified on the manifest but it is likely the activity would have been accounted for. This means the activity for that waste shipment has been included in the Radiological Risk Assessment (Appendix II) that was done for the Final EIS (Thatcher 2003). The results of the Radiological Risk Assessment are discussed in Section 4.4, Post-Closure Radiological Dose.

(from References)

U.S. NRC, 2003, *Long-Term Hazard of Millstone Unit 1's Missing Spent fuel Rods Potentially Disposed at the Hanford Commercial Low-Level Radioactive Waste Disposal Facility,* January 6, 2003, FR Volume 68, No. 3, 588-589.

Washington State Department of Health
Office of Radiation Protection

RADIOLOGICAL RISK ASSESSMENT

LOW-LEVEL RADIOACTIVE WASTE DISPOSAL SITE
RICHLAND, WASHINGTON

Andrew H. Thatcher, DOH

October 2003

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1.0 INTRODUCTION

This report contains the analyses and results for estimating long-term health effects from closing the commercial low-level radioactive waste (LLRW) disposal site in Richland, Washington. The report supports the Final Environmental Impact Statement (FEIS) being prepared by the Washington State departments of Health and Ecology. This report addresses long-term risk from the radiological waste disposed at the site from 1965 through the projected closure date. The objective of this report is to compare the relative long-term risk of the proposed closure plan to the alternatives to that plan (referred to collectively as the "alternatives"). For each alternative, the following analyses have been performed:

- Yearly dose estimates for the post-closure exposure scenarios
- Incremental lifetime cancer risks based on post-closure scenarios
- Predicted impacts to individuals as a result of inadvertent human intrusion

Section 2 briefly reviews the proposed closure plan and the alternatives. Section 3 presents the six exposure scenarios used for the risk calculations. Included in this section is a review of how the scenarios used in this analysis compare to the DOH Hanford Guidance for Radiological Cleanup, the Hanford Site Risk Assessment Methodology (HSRAM), and the State Model Toxics Control Act (MTCA). Section 4 provides a review of the methodology used to calculate the risk.. Section 5 presents the risk results and dose results of the proposed alternatives for the six areas of analysis described in Section 3. Section 6 discusses the uncertainty analysis for the intruder and an offsite individual. Finally, Section 7 contains a summary of the results.

2.0 PROPOSED ALTERNATIVES

The alternatives for the closure of the LLRW disposal site each include a cover over the site. The alternatives were designed to represent a reasonable range of cover designs and closure times. The primary difference is in their ability to stop the infiltration of water to the contaminated waste. Table 2.1 provides a brief synopsis of the different alternatives.

2.1 Description of Alternatives

Table 2.1 Description of Alternatives

Alternative Description	Final Close Date	Cover Description	Cover Infiltration through Top Layers
Proposed Action	Year 2056	Multi-layer cover with 4-inch 50% gravel surface layer, 36-inch silt loam and sand/bentonite infiltration barrier. Site soil layers added for total cover depth of 16' 4".	2 mm/yr
Filled Site -Geomembrane and GCL Layer	Year 2215	Uses Geomembrane and GCL but assumes the site is filled to capacity through accepting higher annual volumes or extending the closure date.	2 mm/yr
Site Soils	Year 2056	Single layer cover of 11 feet of site soils.	20 mm/yr
Thick Homogeneous Cover	Year 2056	Three layer cover with 60-inch silt loam layer. Site soil layer added for total cover depth of 16' 6". No drainage barrier.	0.5 mm/yr
Enhanced Designs: **Design A – Asphalt layer** **Design B – Geomembrane and GCL layer** **Design C – Sand/bentonite layer**	Year 2056	Three cover designs – all have 60 inches of site soil but with different drainage barrier. Each cover has site soil layers added for total cover depth of 16' 6".	0.5 mm/yr
Enhanced Geomembrane and GCL layer - late	Year 2056	Uses Enhanced Geomembrane and GCL cover, but the trenches are not covered until 2056	0.5 mm/yr
Enhanced Geomembrane and GCL layer	Year 2005	Uses Enhanced Geomembrane and GCL cover, but site is closed in year 2005.	0.5 mm/yr

3.0 EXPOSURE SCENARIOS

In order to determine the risk that an individual would be expected to receive from the closure alternatives, scenarios are developed to approximate the lifestyles of the hypothetical individuals. The scenarios used for evaluation of the potential impacts from the LLRW disposal site are:

1. Offsite Rural Resident Scenario

2. Offsite Native American Scenario

3. Intruder Rural Resident Scenario

4. Intruder Native American Scenario

5. Intruder Native American Upland Hunter Scenario

6. Native American Subsistence River Resident

The basis for the general population scenarios can be found by reviewing the environmental impact statements supporting 10 CFR 61 [U.S. NRC, 1981, 1982], as well as the Hanford Site Risk Assessment (HSRAM) manual [U.S. DOE, 1995] and the DOH Hanford Guidance for Radiological Cleanup [DOH, 1997]. A comparison of the parameters defined for this analysis, the HSRAM manual, and the state of Washington Model Toxics Control Act (WAC 173-340) is provided. The Native American Subsistence scenario was modified from the CRCIA document [U.S. DOE 1998] and the Tank Waste Remediation System FEIS [U.S. DOE 1996], following consultation with representatives of the Confederated Tribes of the Umatilla Indian Reservation, the Yakama Indian Nation, and the Nez Perce Tribe. Both the Native American Upland Hunter and Columbia River Subsistence Resident scenario were obtained from the CRCIA document.

3.0.1 Potential Impacts to a Child

Included in the rural resident scenario and Native American scenario is an analysis of the potential impacts to a child. The child scenario is developed using the same exposure pathways as the adult, but utilizes different intake parameters. The consumption information for the children is based upon data from the 1977-1978 Nationwide Food Consumption Survey conducted by the U.S. Department of Agriculture [Callaway, 1992]. The mean value is used as the basis for the consumption rates for nine different food categories.

The incremental lifetime cancer risk for the child is based upon a composite analysis that is evaluated using child parameters for six (6) years, and adult parameters for 24

years. For the six years as a child, the parameters correspond to the average consumption patterns of the 1-4 and 5-9 age groups.

3.0.2 Timing of Scenarios

Upon cessation of activities at the LLRW disposal site, the facility begins a multi-year final closure on those trenches not previously closed. A period of active monitoring begins immediately after final closure activities are complete. This "institutional control" period could last for several centuries,[1] but for this analysis, the active monitoring period is assumed to last only 107 years.[2] During the institutional control period, lapses in land records that would result in inadvertent land purchase and squatting are presumed to not occur. As a result, intruder analysis predicting the impact to individuals of the general population or critical populations does not begin until 107 years following final closure.

It is conceivable for an individual to reside at the LLRW disposal site boundary prior to the end of institutional control.[3] In this event, exposure via a groundwater well or diffusion of radioactive gases could result in an impact during the 107-year institutional control period. In the methodology discussion, the impact of those exposures is included in the H-3, C-14, and Ra-226 discussions.

The following sections provide a description of the scenario, an outline of the pathways analyzed, and tables that indicate the parameters used in the analysis.

3.1 The Adult and Child Rural Resident Scenario: Offsite General Population

The rural resident is an individual living in a remote or sparsely populated area. The individual spends all of his/her time on his/her parcel of land. In order to maximize exposure, the individual resides at the LLRW disposal site boundary in a location that is the predominant downwind and downstream direction. The individual builds a house, drills a well, and raises crops and animals in order to support his/her rural lifestyle. Due to the limitations of the quantity produced and variety of fruits and vegetables, only a portion of the produce is grown on his/her land. Due to the use of the groundwater well, the individual is exposed to a number of pathways. The pathways analyzed for the rural resident scenario are [Kennedy and Strenge, 1992][4]:

- External exposure to radiation from contaminated soil while outdoors

[1] A fund is currently held by the state that has sufficient funds to ensure that active monitoring and maintenance activities can continue well into the future.
[2] 107 years represents 100 years of institutional controls and seven years of onsite "active" maintenance.
[3] The disposal site remains located within the proposed active control area of the 200 Area [Kincaid, et al, 1998]. This active U.S. DOE institutional control would also have to lapse for an individual to reside at the boundary of the disposal site.
[4] Additional pathways that are considered but not analyzed are included in the methodology discussion. Examples are dermal absorption, and inhalation of groundwater contaminants while showering.

- External exposure to radiation from contaminated soil while indoors
- Inhalation exposure to resuspended soil while outdoors
- Inhalation exposure to resuspended soil while indoors
- Inhalation exposure to resuspended surface sources of soil tracked indoors
- Inhalation exposure to gaseous radionuclides while indoors and outdoors
- Direct ingestion of soil
- Inadvertent ingestion of soil tracked indoors
- Ingestion of drinking water from a groundwater well (including while showering)
- Ingestion of plant products grown in contaminated soil
- Ingestion of plant products irrigated with contaminated groundwater
- Ingestion of animal products grown onsite

The offsite analysis assumes that exposures can only result from contaminated groundwater and/or aerial deposition from resuspended contaminated particles driven offsite. Inhalation of gases such as radon can occur through atmospheric dispersion. In the analysis, potential impacts such as resuspension from onsite are assumed to occur as a result of an onsite intruder. Table 3.1.1 provides an overview of the exposure pathways for the rural resident.

Table 3.1.1 Offsite Rural Resident Exposure Pathways

Exposure Pathways	Radionuclide Exposure
External exposure from gamma emitting radionuclides in soil while outdoors	Yes
External exposure from gamma emitting radionuclides in soil while indoors	Yes
Inhalation of resuspended soil and dust	Yes
Inhalation of radon and radon decay products from soil containing radium	Yes
Incidental ingestion of soil	Yes
Ingestion of drinking water transported from soil to potable groundwater sources	Yes
Ingestion of water containing contaminants during showering	Yes
Indoor inhalation	Rn-222 only
Dermal absorption of contaminants via skin or puncture wounds	Tritium only
Ingestion of home grown produce (fruits and vegetables)	Yes
Ingestion of meat containing contamination taken up by cows grazing on contaminated plants	Yes
Ingestion of milk containing contamination taken up by cows grazing on contaminated plants	Yes
Ingestion of meat and eggs containing contamination taken up by poultry feeding on contaminated produce	Yes
Ingestion of locally caught fish	No
Ingestion of organ meats, upland birds, waterfowl, wild bird eggs	No
Ingestion of game meat containing radionuclides	No

Table 3.1.2 compares the exposure parameters for the rural resident to the Agricultural scenario in HSRAM, the rural resident scenario in the DOH guidance document and the available guidance found in MTCA. This comparison is conducted because HSRAM and MTCA are recognized as the governing cleanup approaches at the Hanford Reservation. The DOH Guidance is referenced extensively in cleanup actions.

Significant differences between the rural resident scenario for this EIS and the guidance for HSRAM, DOH Guidance, and MTCA are:

- Soil ingestion rates – HSRAM and DOH Guidance recommends 100 mg/d for the adult; MTCA recommends 50 mg/d. This report uses 50 mg/d. The 50 mg/d is further supported in the extensive soil ingestion review performed by S.L. Simon [Simon, S.L., 1998].

- HSRAM considers dermal exposure and absorption. This analysis considers dermal exposure and absorption only for tritium (dermal absorption is discussed in greater detail in Section 4.3.4), as the absorption fraction for most radionuclides is quite small and not a large contributor to dose. DOH Guidance does not consider dermal absorption.

- HSRAM considers groundwater and surface water inhalation; DOH Guidance does not. Surface water inhalation is not considered for this analysis, as the LLRW disposal site is not near a surface water source. Groundwater inhalation is considered for the Native American sweat lodge scenario. Groundwater inhalation while showering is briefly analyzed in Section 4.2.3 and is determined to not be a significant contributor to dose.

- Sediment ingestion is not considered in this analysis, as no surface water source exists in close proximity.

- The EIS rural resident scenario does consider the ingestion of meat, poultry, eggs, and dairy products that are not considered in MTCA or HSRAM. DOH Guidance considers the ingestion of meat, poultry, and dairy products, but does not consider egg ingestion. The ingestion values for the EIS rural resident scenario are similar to those found in the DOH Guidance. The EIS is more conservative than the DOH Guidance in the ingestion of beef.

- The rural resident scenario does not consider the ingestion of fish and game meat. Fish ingestion is omitted because no source of surface water exists in close proximity to the LLRW disposal site. Game meat is not considered because the only source for contaminant uptake is via groundwater related activities. Farm animals are therefore viewed as always having a greater potential for exposure than game.

- This Radiological Assessment utilizes slightly lower produce ingestion rates as compared to HSRAM or DOH Guidance. The differences are due to the use of NUREG 5512 as the primary reference for the analysis. The differences are well within the uncertainty of the produce intake rates for adults.

Table 3.1.2 Exposure Parameters Comparison for the Rural Resident

Media/Pathway		Exposure Parameters	Rural Resident Scenario	Hanford Guidance[5]	HSRAM	MTCA[6]
			Exposure/Intake/Contact Rate			
Soil	Ingestion	Soil ingestion rate (mg/d) (child)	200	NA	200	200
		(adult)	50	100	100	50
		Exposure frequency (days/year)	365[7]	365	365	ND
		Exposure duration (years) (child)*	6 yr child, 24 yr adult[8]	NA	6	6
		Exposure duration adult (years)	30	30	24	24
		Body weight (kg) (child)	16	NA	16	16
		(adult)	70	70	70	70
	External	External soil exposure frequency (hours/day)	24	19.2[9]	24	ND
		Exposure duration (years)	30	30	30	ND
	Dermal	Dermal soil exposure rate	NC	NC	ND for radioactive	ND
		Exposure frequency	NC	NC	ND	ND
		Exposure duration	NC	NC	ND	ND
		Body weight (kg)* (child)	16	NA	16	ND
		(adult)	70	NA	70	ND
Air	Inhalation	Inhalation rate adult (m^3/d)	20	20	20	20
		Inhalation rate child (m^3/d)	8.8	NA	ND	ND
		Exposure frequency (days/year)	365	292	365	ND
		Exposure duration (years)**	30	30	30	30
Ground-water	Ingestion	Groundwater ingestion rate (L/d)	3	2	2	2
		Exposure frequency (days/year)	365	365	365	ND
	Inhalation	Groundwater inhalation rate (m^3/d)	NC	NC	15	ND
	Dermal	Dermal exposure rate (min)	NC	NC	10	ND
Surface Water	Ingestion	Surface water ingestion (L/d)	NA	NC	2	[10]
	Inhalation	Surface water inhalation (m^3/d)	NA	NC	15	ND
	Dermal	Dermal exposure rate (time)	NA	NC	ND for radioactive	ND
Sedi-ment	Ingestion	Sediment ingestion rate (mg/d) (child)	NA	NC	200	200
		(adult)	NA	NC	100	50
	Dermal	Dermal exposure rate (mg)	NA	NC	ND	ND

[5] Washington Department of Health Hanford Guidance for Radiological Cleanup, 1997, Rev. 1.
[6] MTCA does not provide for pathway analysis; instead, parameters are given in order to calculate a cleanup level in various media. As a result, pathways such as external exposure and the intake of biota (other than fish) are not considered.
[7] Parameters recommended in WAC 173-340-720, WAC 173-340-740, or WAC 173-340-750, Method B, except as noted.
[8] For the child analysis, six years exposure is assumed as a child, and 24 years as an adult.
[9] The Hanford Guidance document breaks down the time spent in the contaminated area to 60% indoors, 20% outdoors, and 20% offsite.
[10] Surface water cleanup levels for MTCA are based upon fish ingestion.

Media/Pathway		Exposure Parameters	Rural Resident Scenario	Hanford Guidance[5]	HSRAM	MTCA[6]
			Exposure/Intake/Contact Rate			
		(child)				
		(adult)	NA	NC	ND	ND
Biota	Dairy	Dairy consumption rate (l/d)	0.27	0.27	300 g/d	ND
		Dairy exposure frequency (days/year)	365	365	365	ND
	Beef	Beef consumption rate (g/d)	162	75[11]	75	ND
		Beef exposure frequency (days/year)	365	365	365	ND
	Game	Game consumption rate (g/d)	0	NC	1	ND
		Game exposure frequency (days/year)	365	NC	365	ND
	Fish	Fish consumption rate (g/d)	0	14.8	54	54
		Fish exposure frequency (days/year)	365	365	365	ND
	Fruit	Fruit consumption rate (g/d)	38	42[12]	42	ND
		Fruit exposure frequency (days/year)	365	365	365	ND
	Vegetable	Vegetable consumption rate (g/d)	68	80	80	ND
		Vegetable exposure frequency (days/year)	365	365	365	ND
	Poultry	Poultry consumption rate (g/d)	25	25	ND	ND
		Poultry consumption frequency (day/year)	365	365	ND	ND
	Eggs	Egg consumption rate (g/d)	27	NC	ND	ND
		Egg consumption frequency (day/year)	365	NC	ND	ND

NC Not Calculated
NA Not Applicable
ND Not Defined
*Body weights are 16 kg for children, and 70 kg for adults.
**Exposure duration is 6 years for children (when ages are specified for children), and 30 years for adults.

3.2 The Native American Scenario: Offsite Critical Population

The general framework surrounding the scenario was borrowed from DOE/EIS-0189, *Final Environmental Impact Statement for the Hanford Tank Waste Remediation System* [U.S. DOE, 1996]. This scenario combines both traditional and contemporary lifestyles. The traditional activities are hunting, fishing, and gathering plants and materials. Contemporary activities include the use of groundwater for drinking, showering, and watering for plants and animals. The Native American is assumed to live offsite while using the surrounding area for a variety of the activities.

The Native American scenario represents exposures received during a 70-year lifetime by an individual who engages in both traditional lifestyle activities (e.g., hunting and

[11] Combined with poultry consumption.
[12] Combined with fruits, vegetable, and grain consumption.

using a sweat lodge) and contemporary lifestyle activities (e.g., irrigated farming). The individual is assumed to spend 365 days per year on the LLRW disposal site over a 70-year lifetime. Some activities are assumed to continue year-round, while others are limited by climate (e.g., frost-free days).

The main exposure routes via the groundwater pathway are shown in Table 4.2.1. They are drinking water, consumption of irrigated vegetables and animal products, ingestion of irrigated soil, external exposure to soil contaminated with irrigation water, inhalation of resuspended soil, and inhalation of water vapors in the sweat lodge.[13]

Table 3.2.1 Native American Exposure Pathways

Exposure Pathways	Radionuclides
External exposure from gamma emitting radionuclides in soil while outdoors	Yes
External exposure from gamma emitting radionuclides in soil while indoors	Yes
Inhalation of resuspended soil and dust	Yes
Inhalation of radon and radon decay products from soil containing radium	Yes
Incidental ingestion of soil	Yes
Ingestion of drinking water transported from soil to potable groundwater sources	Yes
Ingestion of water containing contaminants during showering	Yes
Indoor inhalation	Rn-222 Only
Dermal absorption of contaminants via skin or puncture wounds	Tritium Only
Ingestion of home-grown produce (fruits and vegetables)	Yes
Ingestion of meat containing contamination taken up by cows grazing on contaminated plants	Yes
Sweat Lodge Inhalation	Yes
Ingestion of milk containing contamination taken up by cows grazing on contaminated plants	Yes
Ingestion of meat and eggs containing contamination taken up by poultry feeding on contaminated produce	Yes
Ingestion of locally caught fish	No
Ingestion of organ meats, upland birds, waterfowl, wild bird eggs	Yes
Ingestion of game meat containing radionuclides	Yes

Parameters for the Native American scenarios were derived from Harris and Harper [Harris and Harper, 1997], with supplemental information from the TWRS [U.S. DOE, 1996] and CRCIA [U.S. DOE, 1998] analyses. Ingestion rates of native foods are based on surveys cited in Harris and Harper. The EPA vegetable ingestion rate was ratioed into "root" and "leafy" by the proportions referenced from Hunn [Hunn, 1990]; i.e., 1300 g/d roots and 1400 g/d other vegetables for a total of 2700 g/d vegetables. Ingestion of animal organs and wild bird meat was accounted for by increasing the total meat and poultry intake rate. Animal organs were assumed to have contaminant concentrations 10 times the concentration of other tissues, and the organ intake rate was assumed to be 10 percent of the intake rate of other animal tissue.[14] Note,

[13] As discussed in Section 4.2.4, groundwater inhalation while showering is shown to not significantly contribute to dose.

[14] The assumption of 10 times the concentration in organ meats is over-conservative for most radionuclides of interest for the groundwater. Cl-36 distributes itself uniformly in the body, so no tissue or organ concentration is enhanced. Tc-99 has an overall organ (GI tract, kidneys, and liver) concentration

however, that ingestion of animal products is unlikely to be a significant pathway. Buried waste must be brought to the surface for it to have any effect on the wild animal population. Contaminated waste which is brought to the surface would be distributed in a limited area, small in comparison to the home range of the animal. Exposure times for soil were assumed to last 12 hours a day for 365 days, or 180 days/year for 24 hours. Table 3.2.2 shows the exposure parameters specific for the Native American scenario.

The Native American scenario represents the use of a subsistence Native American lifestyle that includes contemporary activities such as irrigated agriculture, as well as activities such as hunting and the gathering of plants and materials.

Table 3.2.2 Exposure Parameters Comparison for the Native American

		Native American-Specific Exposure Parameters	EIS LLRW disposal site Scenario	TWRS	CRCIA	Harris and Harper
Media	**Pathway**	**Exposure Route**	**Intake/Contact**			
Soil	Ingestion	Soil ingestion rate adult and child (mg/d)	200	200	200	200
		Soil exposure frequency (d/yr)	180	365	365	180
		Exposure duration child (yr)	6	6	ND	ND
		Exposure duration adult (yr)	70	64	70	70
		Body weight child (kg)	16	16	ND	ND
		Body weight adult (kg)	70	70	70	70
	External	External exposure time soil (h)	24	24	24	24
		Soil exposure frequency (d/yr)	180	365	365	180
		Exposure duration adult (yr)	70	64	70	70
		External shielding factor	0.8	0.8	0.8	0.8
	Inhalation	Inhalation Rate - child (m^3/d)	8.76	15	ND	ND
		Inhalation Rate - adult (m^3/d)	30	30	30	20
		Soil exposure frequency (d/yr)	180	365	365	180
		Exposure duration child (yr)	6	6	ND	ND
		Exposure duration adult (yr)	70	64	70	70
		Mass loading g soil/m^3 air)	F(activity)	$1.0x10^{-4}$	$1.0x10^{-4}$	$1x10^{-5}$
Water, food		Fruit ingestion rate (g/d)	231	330	330	231
		Vegetable ingestion rate (g/d)	343 (165 root + 178 leafy)	330	330	343
		Meat ingestion rate (g/d) This includes organ meats at 10 times the meat concentration, and consumed at 0.1 frequency of meat. (animal protein, organs, upland birds, waterfowl, wild bird eggs)	275 (250 meat + 25 organ)	341	337	250 (250 meat + 25 organ)
		Milk ingestion rate (L/d)	.49	0.6	0.6	0.49
		Food ingestion duration (year)	70	70	70	70
		Food ingestion frequency (d/yr)	365	365	365	365

about three times greater than the muscle tissue. I-129 deposits in the thyroid only with the remaining fraction (about 70%) being directly excreted, so no enhanced concentration would likely be found.

		Native American-Specific Exposure Parameters	EIS LLRW disposal site Scenario	TWRS	CRCIA	Harris and Harper
Media	Pathway	Exposure Route	Intake/Contact			
		Water ingestion rate - child (L/d)	1.96	1.5	ND	ND
		Water ingestion rate - adult (L/d)	4.01	3	3	3
	Inhalation	Sweat lodge Water Use rate (L/h)	4		4	4
		Sweat lodge Equivalent hemisphere Diameter (m)	3.05			2
		Sweat lodge exposure rate (h/d)	1	1	1	1
		Sweat lodge frequency rate (d/yr)	365	365	365	365
		Inhalation Rate - child (m^3/d)	15	15	ND	ND
		Inhalation Rate - adult (m^3/d)	30	30	30	20
Air	Inhalation	Inhalation Rate - child (m^3/d)	15	15	ND	ND
		Inhalation Rate - adult (m^3/d)	30	30	30	20
		Inhalation exposure (h/d)	24	24	24	24
		Inhalation frequency (d/yr)	365	365	365	365

ND Not Defined

NOTE: Child parameters for food intake for the Native American are based upon the relative fraction of rural resident child intake, as compared to the rural resident adult. This fraction is then multiplied by the Native American adult to obtain the child intake rate for the Native American child.

Included as part of the table for the Native American parameters is a comparison of the exposure parameters recommended in the Tank Waste Remediation System (TWRS) EIS [U.S. DOE, 1996], the Columbia River Comprehensive Impact Assessment [U.S. DOE, 1998], and the Harris and Harper guidance on Native American Subsistence. A review of the table indicates that when differences between the three references exist, the Harris and Harper document is used as the default. The one exception to this is the decision to use a 30-m^3/day inhalation rate as opposed to 20 m^3/day.[15]

The Native American Sweat Lodge

Use of a sweat lodge is unique to the Native American scenario. The sweat lodge is similar to a steam bath, where high temperatures are combined with a humid environment. The potential ability of the liquid contaminants to become airborne during the flashing of the water to steam on the rocks of the sweat lodge makes this portion of the scenario of particular importance, as the radiological impact of an inhaled contaminant far exceeds the radiological impact of a similar quantity of an ingested contaminant.[16] The Native American adult is assumed to spend 1 hour/day in a sweat lodge.

[15] The inhalation rate change is based upon a request by Stuart Harris, Confederated Tribe of the Umatilla Indian Reservation.

[16] Briefly, as the steam is vaporized on the hot rocks, liquid droplets are propelled out with the steam. These liquid droplets have not fully transitioned to steam yet. This has an impact for the air concentration calculated for a given volume and temperature, as the steam tables would not take into consideration the liquid droplets. The contaminants of interest for the groundwater are not volatile for the temperatures of concern in a sweat lodge.

To briefly describe some of the central parameters of a sweat lodge, the temperature ranges anywhere from 120° to 200° F[17]. Approximately one gallon of water is used per hour. The water that is used to create the steam is heated prior to application on the rocks. The rocks are rotated from the fire to ensure that they stay hot. Estimated temperature of the rocks is 500°F to 600°F.

Children are known to also participate in the sweat lodge, although their time spent is less frequent and the duration is only 10-15 minutes. It should also be noted that it is common for elders to participate in sweat lodges several times a day for hours at a time. For the Native American adult, an additional liter of water[18] is assumed to be consumed during their time in the sweat lodge to account for the water loss due to sweating.

3.3 The Rural Resident Intruder Scenario

Section 3.0.2 discussed the concept of institutional control, which prevents living on the LLRW disposal site. Should there be a lapse of institutional controls, an individual may accidentally live on the site without the knowledge that she/he is residing on the LLRW disposal site. Although significant impediments are in place to ensure that such an intruder condition does not occur, the intruder scenario is designed to estimate the dose to such an individual. The intruder analysis is in direct contrast to an individual who intentionally lives on the LLRW disposal site, disregards site markers, and removes or uncovers contaminated waste.

The onsite intruder, rural resident requires a well in order to live, grow crops, and feed livestock in an arid climate. This scenario is identical to the offsite rural resident with the single exception that, when drilling the well, the onsite intruder removes contaminated well cuttings to the surface. This scenario identifies and quantifies the dose estimate as a result of bringing the well cuttings to the surface, and adds this to the exposure as a result of using the contaminated well water (see Section 3.1, the offsite rural resident). The pathways of exposure for the intruder are similar to the irrigation pathways for the rural resident and include contaminated plant ingestion, soil ingestion and inhalation (via resuspension), and external radiation from the contaminated soil. The ingestion of animal products further contaminated from well cuttings is not assumed, as the limited amount of contaminated material can at best only be spread to an area of 1000 to 2000 m² [U.S. NRC, 1981].[19] The animals are, however, potentially contaminated as a result of the use of irrigation water. The area of the contaminated material distributed on the surface is conservatively assumed to sufficiently encompass the perimeter of the house, thereby contributing to an indoor dose from external radiation.

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[17] 75 degrees C (~170F) is the average temperature assumed for the water concentration in the air.
[18] The additional water intake is corrected from 2 L/d during the sweat to an additional 1 L/d for a 1hour sweat.
[19] The contribution of dose to humans from animals, were they to be included in the dose estimate, would have a contribution similar to that of the plant contribution (<1%).

The adult rural resident intruder is assumed to spend all of his/her time on the LLRW disposal site, 60% of which is spent indoors and 40% outdoors. Of the time spent outdoors, 60% (of the total 2,500 m^2) is assumed to be spent within the assumed 1,500 square meter surface contaminated area.[20] In the case of individuals from six to 20 years of age, time is allocated for attending school. The school attendance time is assumed to take away from the time that children spend outdoors, leaving the indoor time for children the same as for the adult. The remaining outdoor time for the children ages 6 to 20 years is assumed to take place within the 1,500 square meter surface contaminated area.

The exposure pathways and parameters for the rural resident intruder scenario are the same as for the offsite rural resident. However, the source term is significantly larger (see the source term discussion in Section 3 for a list of specific contaminants), as the intruder is exposed to a greater quantity of radioactive contamination. The offsite intruder, by comparison, is only directly exposed to the contaminated waste as a result of irrigation and diffusion and resuspension from intruder activities.

3.4 The Native American Intruder Scenario

The Native American intruder scenario utilizes the same exposure parameters as the offsite Native American scenario. The Native American intruder assumptions for access to the buried waste are identical to the intruder rural resident. Please refer to the pathways and parameters located in Tables 3.2.1 and 3.2.2, and the intruder waste removal discussion in Section 3.3 for review.

3.5 Intruder Scenario: The Upland Hunter

The general operating assumption for a revised intruder scenario is that U.S. DOE's central plateau's institutional controls never lapse [U.S. DOE, 1999]. Considering that the lands in the Central Plateau will remain in use for the management of radioactive and hazardous waste from multiple sources, it is more realistic (while still conservative) to consider the onsite intruder as an individual that would not live on the site but instead inadvertently enters the Central Plateau for a limited period of time. Given the continued management of the Central Plateau, the Native American Upland Hunter [U.S. DOE, 1998] would be considered a reasonable maximum exposure (RME). This approach is consistent with the approach for loss of institutional controls at MTCA sites.

This scenario could result in exposures via the ingestion of meat (game), the ingestion of plants/roots, inhalation of radon, C-14 and tritium, and groundwater ingestion.[21] Although the hunter is assumed to bring drinking water to the site that is contaminated from site operations, the hunter is not assumed to bring sufficient water for use in a

[20] If the contaminated material were spread over 2,500 square meters, the external dose estimate would remain the same, as the concentration would decrease by a commensurate amount.
[21] The water is carried to the site by the hunter and is conservatively assumed to be from a source of water that is contaminated from the LLRW site.

sweat lodge while hunting.[22] No direct contact with the waste by a hunter is assumed, as the water is greater than 16' in depth.[23] As a result, the direct ingestion of contaminated soil and external exposure are not pathways considered in the FEIS. The meat and plant ingestion pathways are only considered in light of their uptake of C-14 and tritium as a result of gaseous diffusion through the soil cover.

The main exposure routes are shown in Table 3.5.1.

Table 3.5.1 Upland Hunter Exposure Pathways

Exposure Pathways	Radionuclides
External exposure from gamma emitting radionuclides in soil while outdoors	No
External exposure from gamma emitting radionuclides in soil while indoors	No
Inhalation of resuspended soil and dust	No
Inhalation of radon and radon decay, Tritium, C-14 products while outdoors	Yes
Incidental ingestion of soil	No
Ingestion of drinking water transported from soil to potable groundwater sources (from offsite source)	Yes
Indoor inhalation	No
Dermal absorption of contaminants via skin or puncture wounds	Tritium Only[24]
Ingestion of Native Plants	Yes
Sweat Lodge Inhalation	No
Ingestion of locally caught fish	No
Ingestion of organ meats, upland birds, waterfowl, wild bird eggs	Yes
Ingestion of game meat containing radionuclides	Yes

Parameters for the Native American scenarios were derived from Harris and Harper [Harris and Harper, 1997]. Ingestion rates of native foods are based on surveys cited in Harris and Harper. The EPA vegetable ingestion rate was ratioed into "root" and "leafy" by the proportions referenced from Hunn [Hunn, 1990]; i.e., 1300 g/d roots and 1400 g/d other vegetables for at total of 2700 g/d vegetables. Ingestion of animal organs and wild bird meat was accounted for by increasing the total meat and poultry intake rate. Animal organs were assumed to have contaminant concentrations 10 times the concentration of other tissues, and the organ intake rate was assumed to be 10 percent of the intake rate of other animal tissue.[25] Note, however, that ingestion of animal products will not be a source of contamination, as the contamination depth is too great to be accessible by humans, plants, or animals. Table 3.5.2 shows the exposure parameters specific for the Native American scenario.

[22] At least in the current environment, the Central Plateau of the Hanford Site lacks sufficient vegetation with which to build a sweat lodge.

[23] Aside from human intrusion, potential biotic intrusion was evaluated in Section 4.3.5 of Appendix I of the DEIS. In summary, no native plant or animal burrows to the depth of the contaminated material.

[24] Further discussed in Section 4.3.4 of Appendix I of the DEIS.

[25] The assumption of 10 times the concentration in organ meats is over-conservative for most radionuclides of interest for the groundwater. Cl-36 distributes itself uniformly in the body, so no tissue or organ concentration is enhanced. Tc-99 has an overall organ (GI tract, kidneys, and liver) concentration about three times greater than the muscle tissue. I-129 deposits in the thyroid only, with the remaining fraction (about 70%) being directly excreted, so no enhanced concentration would likely be found. Uranium and plutonium are bone seekers but will also deposit a fraction to the kidneys.

The Native American scenario represents the use of a subsistence Native American lifestyle that includes activities such as hunting and the gathering of plants and materials.

Table 3.5.2 Exposure Parameters Comparison for the Native American

		Native American-Upland Hunter Exposure Parameters	FEIS
Media	Pathway	Exposure Route	Intake/Contact
		Exposure Frequency	24 hr/d
		Exposure Duration	7 d/y
		Body weight child (kg)	16
		Body weight adult (kg)	70
Soil	Ingestion	Soil ingestion rate adult and child (mg/d)	200[26]
	External	External exposure time soil (h)	24
		Soil exposure frequency (d/yr)	7
		External shielding factor	0.8
		Mass loading g soil/m^3 air)	0
Water, food		Fruit ingestion rate (g/d)	231 Adult 127 Child
		Vegetable ingestion rate (g/d)	343[27] Adult 187 Child
		Meat ingestion rate (g/d). This includes organ meats at 10 times the meat concentration, and consumed at 0.1 frequency of meat (animal protein, organs, upland birds, waterfowl, wild bird eggs).	348[28] for 10.44 Kg total for Adult, 212 g/d for 6.4 Kg total for Child
		Water ingestion rate - child (L/d)	2
		Water ingestion rate - adult (L/d)	3.0
Air	Inhalation	Inhalation Rate - child (m^3/d)	15
		Inhalation Rate - adult (m^3/d)	30

NOTE: Child parameters for food intake for the Native American are based upon the relative fraction of rural resident child intake, as compared to the rural resident adult. This fraction is then multiplied by the Native American adult to obtain the child intake rate for the Native American child.

3.6 The Columbia River Scenario: Native American Subsistence River Resident

The Subsistence River Resident Scenario represents a Native American living a traditional lifestyle for 70 years near the Columbia River, on what is now the U.S. DOE Hanford Reservation. The individual, as an adult and as a child, spends time at the river shoreline, at river seeps and springs, as well as in upland areas away from the Columbia River. The Native American individual drinks water from the seeps, bathes

[26] The contaminated soil, at a depth of 16+ feet, is not accessed by humans, plants or animals.
[27] 165 root +178 leafy).
[28] Sufficient meat is assumed to be obtained over a 7-day period to last for 30 days.

and swims in the river, and uses a sweat lodge by the river, using seep water. The individual consumes plant and animal products from the river, the springs, and from the upland areas. Some of the plant foods are irrigated with river water containing radionuclides carried into it from the seeps. The dietary meat includes game and pastured livestock, including organs.

The pasture for the livestock is irrigated with river water containing radionuclides carried into it from the seeps[29]. He or she also gathers and uses materials for cultural purposes from the shoreline, from the springs, and from the upland areas. A more complete list of the sources of exposure considered, is given in Table 3.6.1. The parameter values are listed in Table 3.6.2. This scenario is essentially that used by U.S. DOE in their CRCIA document [U.S. DOE, 1998].

The major change by DOH in this assessment of the parameter values used by U.S. DOE is that the seeps are assumed to be contaminated from groundwater from the commercial low-level waste facility instead of from the Hanford reservation itself. The concentrations in the seeps are assumed to be diluted 53% by river water [Guensch, G.R & Richmond, M.C., 2001]. Another important modification from the U.S. DOE assessment is that the only significant source of potential contamination away from the seeps in the upland areas is from irrigation using seep water. Animals obtained upland, are themselves potentially contaminated only from foraging on the crops and are thus not likely to be contaminated to any measurable extent. They are not directly contaminated from soils unless those soils are contaminated as a result of irrigation water used from seeps.

The most important assumption of this Columbia River scenario for the Native American Subsistence River Resident is that the seeps are conservatively assumed to have as their source the groundwater that has passed below the low-level waste facility. Thus the seeps are assumed to have the same level of contamination as the groundwater immediately down gradient from the site. This simplifying assumption is extremely conservative, as it does not allow for mixing during the several miles the groundwater travels between the site and the river, nor does it allow for decay during that time period of travel.[30] With this simplifying assumption, neither the parameter "distance traveled" nor the parameter "time period for travel and decay" is used.

[29] For simplicity, the animals are assumed to drink from water at the same concentration as the seeps.
[30] Long-lived radionuclide activities would not decrease significantly during this travel time period in any case.

Table 3.6.1 Native American Subsistence River Resident

Potential Exposure Pathways	Included	Radionuclides
External exposure from gamma emitting radionuclides in soil while indoors	No	
Inhalation of resuspended soil and dust	No	
Inhalation of radon and radon decay, tritium, C-14 products while outdoors	Yes	As a result of tritium and C-14 in the groundwater
Incidental ingestion of soil[31]	Yes	
Ingestion of drinking water transported from soil to potable groundwater sources (from offsite source)	Yes	
Indoor inhalation	No	
Dermal absorption of contaminants via skin or puncture wounds	Yes	Tritium only[32]
Ingestion of native plants	Yes	
Sweat lodge inhalation	Yes	
Ingestion of locally caught fish	No[33]	
Ingestion of organ meats, upland birds, waterfowl, wild bird eggs	Yes	
Ingestion of game meat containing radionuclides	Yes	

Table 3.6.2 Exposure Parameters Comparison for the Native American Subsistence River Resident

		Native American Subsistence Resident Exposure Parameters	FEIS parameter values
Media	**Pathway**	**Exposure Route**	**Intake/Contact**
		Exposure Frequency	24 hr/d
		Exposure Duration	365 d/y
		Body weight child (kg)	16
		Body weight adult (kg)	70
Soil	Ingestion	Soil ingestion rate adult and child (mg/d)	200
	External	External exposure time soil	24

[31] The soil contamination is only as a result of contaminated seep water used for irrigation.
[32] Further discussed in Section 4.3.4 of Appendix I of the DEIS.
[33] Due to the limited volume of seeps as compared to the Columbia River, the fish are not likely to be contaminated to any measurable extent and will therefore not be included in the quantitative analysis.

		Native American Subsistence Resident Exposure Parameters	FEIS parameter values
Media	**Pathway**	**Exposure Route**	**Intake/Contact**
	Inhalation	Air mass loading (ug/m3)	100
Water, food	Ingestion	Fruit ingestion rate (g/d)	231 Adult 127 Child
		Vegetable ingestion rate (g/d)	343[34] Adult 187 Child
		Meat ingestion rate (g/d) This includes organ meats at 10 times the meat concentration, and consumed at 0.1 frequency of meat (animal protein, organs, upland birds, waterfowl, wild bird eggs).	348 g/d for Adult, 212 g/d for Child
Air	Inhalation	Inhalation Rate - child (m^3/d)	15
		Inhalation Rate - adult (m^3/d)	30
Seep/Spring Water	Ingestion	Water ingestion rate - child (L/d), Water ingestion rate - adult (L/d)	2 3
	Dermal exposure(a)	1 hr/day – tritium only considered	20,000 cm2
Biota(f)	Fruit and vegetation	Ingestion	660 g
	Animal protein(b)	Ingestion	150 g
	Other Organs(c)	Ingestion	54 g
	Milk	Ingestion	0.6 L
	Upland Birds	Ingestion	18 g
	Waterfowl	Ingestion	70 g
	Wild bird eggs	Ingestion	45 g
	Dermal	1 hr/day	20,000 cm2
Cultural (d)	Inhalation	1 hr/day	0.1 L/m3

NOTE: Child parameters for food intake for the Native American are based upon the relative fraction of rural resident child intake, as compared to the rural resident adult. This fraction is then multiplied by the Native American adult to obtain the child intake rate for the Native American child.

(a) The dermal exposure is only considered during periods within the sweat lodge.
(b) The animal protein consumption rate includes meat, fat, and marrow, prepared fresh or dried. The equivalent fresh weight is given here.
(c) Approximated as 10 percent of the fish ingestion value.

[34] 165 root +178 leafy).

(d) The unique pathway related to volatilization of contaminants from water during sweat bathing is included here. The absolute humidity is based on saturated conditions at a temperature of 70 to 80 degrees Celsius (160 to 180 degrees Fahrenheit).

4.0 DOSE/RISK ANALYSIS METHODOLOGY

This section describes the methodology used to calculate impacts for the general population, Native Americans, and construction individuals. The discussion of the methodology is divided into the exposure pathways. The pathways are:

- Groundwater
- Soil
- Air
- Food
- Surface water

Food is included as a separate exposure pathway even though contamination of food products actually occurs through water, soil, and air contamination. The food pathway was separated so its impact was clearly shown.

The analysis supporting the dose and risk calculations is applied to all scenarios by changing the parameters or slightly modifying an equation. For brevity, the onsite analysis refers to the intruder analysis. The calculations supporting the ingestion and inhalation pathways are borrowed in part from Kennedy and Strenge [Kennedy and Strenge, 1992]. Calculations for the radon pathway are obtained, with a few modifications, from NRC Reg Guide 3.64 [U.S. NRC, 1989] and the RESRAD manual [Yu, et al, 1993]. The carbon 14 diffusion estimates, although a small contributor to dose, are derived by Dr. Man-Sung Yim [Yim, 1997], with the supporting dose calculation methodology taken from RESRAD [Yu, et al, 1993]. Finally, external dose estimates utilized Federal Guidance Report #12 [Eckerman and Ryman, 1993] and the MICROSHIELD computer code [Grove Engineering, 1998].

The dose calculations contained in this report are intended to represent the maximally exposed individual (MEI) for the rural resident analysis, generally taken to imply the upper 95% confidence interval on the mean, and the average exposure of the critical group, the Native American. All of the calculations are performed using a single-point dose estimate. The assumptions supporting the single-point estimates are conservative and are intended to ensure that the dose projections are sufficiently protective of human health. Uncertainty analysis is performed on the dose projections in Section 6.

The conversion of the estimated dose to risk is performed using the recommended value from ICRP 60 [ICRP, 1990]. This value, 0.0005/Rem for the general population, is a widely applied fatality coefficient and should allow for comparison of radiological risk with other studies.

Modeling Assumptions

The assumptions supporting the groundwater analysis are provided in the Groundwater Analysis Section of this FEIS. Among other items, the groundwater section outlines the infiltration estimates for the various covers, the specific parameters assumed for each radionuclide, and the assumptions used in determining the source term for the groundwater analysis. Source term assumptions are provided in Section 4.1 that follows. Other assumptions used in the analysis of the impacts to individuals are included in the specific sections discussed throughout Section 4 but are briefly outlined below:

- All source term is disposed of at the waste site on the first day of operations, and covered immediately with a final cover. This assumption conservatively places source term at the site for a longer period but does not take into account the 40+ years that the waste is in place without a final or low infiltration cover. The exception to this assumption is the year 2056 Enhanced Late cover, which assumes that a final or interim cover will not be applied until closure, thereby allowing for a significantly greater infiltration rate.

- The source term was segregated into pre-2005 waste and post 2005 waste. During analysis it was determined that the pre-2005 waste contains a greater concentration of radium and other LLRW radionuclides. Analysis for the various alternatives assumes that the intruder locates in the pre-2005 waste area and receives a slightly greater exposure as a result. Supporting information for this assumption is located in Section 5.

- For radium, a source term audit was performed to determine the depths that various radium wastes were buried. The analysis determined that the depth for radium disposal was primarily determined by the year disposed, and as such, one is able to accurately determine the depth below grade for the various types of radium waste. This correction had a tremendous impact on the radon flux as compared with the analysis performed in the DEIS for this LLRW.

- For all analysis with the exception of radon, no credit is given to container integrity. The lifetime of a typical 55-gallon carbon steel drum is expected to be about 30 years [Yim, 1997] and would serve to limit both the production of gases and the infiltration of contaminants to the groundwater. For radon analysis, no emanation is assumed from sealed radium sources (typically encased within concrete) for 500 years.

- Institutional controls are assumed to exist on the site for 107 years. This includes seven years of active maintenance that follows once the site is closed. Institutional controls of only 100 years for the disposal facility is conservative due to the location of the site within the U.S. DOE complex, and the fact that the maintenance fund for this disposal site is sufficiently large to ensure monitoring indefinitely.

- The food and animal pathway analysis is based upon a non-recycling model. Specifically, the contaminated groundwater that is used for irrigation is applied for scenarios that occur at the end of the groundwater modeling (once the groundwater is contaminated) and are not used as the basis or source of infiltration water. The non-recycling model is used because of the amount of time the site is in existence prior to the assumed lapse of institutional controls, and due to the limited probability of multiple generational intruders on the site, considering its location within the overall Hanford Site.

- The rural resident and Native American intruder on the site are assumed to drill a well through a trench contacting the waste. This is a conservative assumption because there is a substantial area on the site that contains no waste, and the waste must be sufficiently degraded so as not to be identifiable. This assumption is also conservative as it is possible that an intruder would not come into direct contact with the waste. The Native American Upland Hunter scenario does not assume direct contact with the waste.

Barrier Performance Analysis

The covers used in the alternatives represent a wide range of possible designs. The enhanced designs in particular provide an additional measure of safety for both infiltration and gaseous diffusion. Specific assumptions used in the analysis of gas emanation from the waste volume, predominately for radon analysis, are outlined as follows:

- The three enhanced barriers are: a bentonite clay mixture layer 30 cm in thickness; a modified asphalt layer; and a GeoSynthetic cover (HDPE) sandwiched with a GeoSynthetic clay liner (GCL). In the first 500 years of performance, the modified asphalt and GeoSynthetic covers are expected to perform almost perfectly in limiting the emanation of radon gases. Following 500 years, the modified asphalt cover and the GeoSynthetic covers are expected to degrade in performance but essentially remain somewhat comparable to the performance of the bentonite layer for the 500 to 1,000-year timeframe.

- A clay barrier performance varies depending upon a number of conditions, such as the moisture content, clay content in the barrier, type of clay, etc. The diffusion coefficient for the clay barrier is based upon the use of an empirical formula developed by Rogers and Nielson [Rogers and Nielson, 1991] as well as the clay material properties as defined in RAETRAD, a software code developed by Rogers & Associates [Nielson, et al, 1993].

4.1 Source Term

This risk assessment is based on a source term that was calculated from disposal manifests, beginning in 1965 through 1996 [Thatcher and Elsen, 1999]. The source term for the analysis includes all radioactive waste disposed at the site, including both

low-level and NARM waste. The source term does not include chemical waste. Future projections for low-level and NARM waste were based on the 1993 through 1996 disposal volumes and the source term expected from the disposal of the Trojan and Washington Public Power Supply reactor vessels. Use of the source term for the risk assessment required certain assumptions or screening tools. These are:

- The total LLRW disposal site inventory contains about 622 separate isotopes. A majority of these radionuclides are short-lived or of minimal activity. In order to focus the analyses on the radionuclides with the highest likelihood of contributing to dose, screening tools/assumptions were developed. The first screening tool assumes that any isotope with a half-life of less than 5.5 years cannot contribute to dose when the institutional control of 107 years is considered. This screening tool is based on the assumption that the institutional control will be effective at keeping people off the LLRW disposal site for at least 107 years. This first assumption specifically excludes any impact from all radionuclides with half-lives less than that of cobalt 60, including cobalt 60.

  As an example, the 1996 undecayed activity of Co-60 is 552,683 curies. Reducing this activity by 107 years of decay would be calculated as follows:

  Equation 1

$$FinalCobaltActivity = 552{,}683Ci * e^{-(\frac{.693}{5.27} * 107\,years)} = 0.43Ci$$

  The resulting activity of Co-60 107 years later is approximately 0.4 curie, which does not take into consideration the significant amount of decay that occurred prior to 1996.

- The second series of screening tools/assumptions excludes radionuclides with total activities less than 1 curie in 1996. The basis for this assumption relates to the equivalent calculated concentration for a given radionuclide. In order to simplify the impact from uncovering and or removing contaminated waste from a buried trench, the LLRW disposal site is assumed to be one homogeneous waste volume. Taking this homogenous waste volume of the actual trenches (not the volume between the trenches), and assuming a waste density of 1.26 g/cm^3 [U.S. Ecology, 1996], results in a total waste mass, including fill, of approximately 1.4x10^{12} g of waste material. Taking a 1-curie source, which is 1x10^{12} pCi, and dividing by the total waste mass, results in a concentration of less than 1 pCi/g. For conservancy, Nb-94, with a total 1996 activity of 0.98 curie, is included in the analysis.

- Decay of radionuclides is considered, as is progeny ingrowth.[35]

4.1.1 Source Term Considerations for Groundwater Modeling

[35] Radionuclides included in the 1965-1996 source term are not decayed prior to 1996. The 1965-1996 source term is decayed as of 1996. All projections of future activities are decay corrected.

Of the total 600+ radionuclides disposed at the LLRW disposal site, very few have a long enough half-life, large enough source term, and are soluble enough to cause a potential impact to groundwater. The radionuclides that are considered in the groundwater analysis are H-3, C-14, Cl-36, Tc-99, I-129, U-234, U-235, U-238, Pu-238, and Pu-239 [Rood, A.S., 2003].

4.1.2 Radionuclides with Source Term Uncertainty

There are two radionuclides with known source term errors. Those radionuclides are Tc-99 and I-129. The Tc-99 and I-129 error is due to the reported activity being based upon scaling factors (the ratio between the difficult-to-detect I-129 and a readily measurable isotope such as Co-60). In actual practice, the minimum detectable activity (MDA) of I-129 and Tc-99 was used for the calculation of the scaling factor and resulted in overestimates of the actual quantities of I-129 by anywhere from 100 to 10,000 [U.S. NRC, 1996]. As is discussed in the Groundwater Appendix, this potential error has little impact on the predicted total dose from groundwater.

4.2 Groundwater

Groundwater contamination has the potential to impact the greatest number of individuals. The primary route for exposure to individuals is direct ingestion of groundwater used as drinking water. Other avenues for exposure include exposure via inhalation and ingestion while showering, or inhalation while in steam rooms, as is the case for the Native American sweat lodge. The use of contaminated groundwater also impacts a number of other pathways, such as soil. The combination of the water and resulting soil contamination, as is the case for the use of groundwater in irrigation scenarios, can also impact food and animal products. This, in turn, may lead to potential exposures to individuals. Please refer to the groundwater section of this EIS for further discussions of the groundwater analysis used in estimating the contaminant concentration. The groundwater concentration estimates for the various alternatives are included in Table 4.2.1.

Table 4.2.1 Summary of Predicted Groundwater Concentrations for the Alternatives* (pCi/l)

Radionuclide	Alternatives							
	Proposed Action	Filled Site	Site Soils Cover	Thick Homog-eneous Cover	Enhanced Asphalt	Enhanced Geo-Synthetic	Enhanced Bentonite – Year 2056	Enhanced Bentonite - Year 2000
Chlorine 36	36	38	45	20	20	20	20	19
Technetium 99	490	590	580	270	270	270	270	250
Iodine 129	3.9	4.5	4.6	1.9	1.9	1.9	1.9	1.8
Uranium 235	0.23	0.23	2.3	0.057	0.057	0.057	0.057	0.057
Uranium 238	0.036	0.036	0.36	0.0089	0.0089	0.0089	0.0089	0.0089

*Estimates are only shown for those radionuclides that are expected to reach the groundwater in less than 10,000 years.

4.2.1 Groundwater Ingestion

Adults in a rural resident scenario are assumed to drink three liters of water per day[36]. Native Americans are assumed to drink five liters of water per day. The two additional liters are due to the additional water use during their time in the sweat lodge. Children for either scenario are assumed to drink a quantity that is a function of their age. The formula for calculating the drinking water dose is as follows:

Equation 2

$$Dose_{dw} = \frac{C_w}{27} * Q_w * DCF * 10^5$$

Where:

- $Dose_{dw}$ = Committed effective dose from drinking water (mrem/year)
- C_w = Contaminant groundwater concentration (pCi/l)
- Q_w = Intake rate of water (l/year)
- DCF = 50 year committed effective dose conversion factor for ingestion of contaminants (Sv/Bq)[37]
- 10,000 = Converts Sieverts (Sv) to mrem
- 27 = Converts Bq to pCi

4.2.2 Groundwater Inhalation: Sweat Lodge

The sweat lodge for the Native American assumes that all the water (and contaminants) used is vaporized or entrained in the lodge, and the resulting concentration breathed for the entire duration in the lodge. The formula for calculating the exposure is:

Equation 3

$$Dose_{sweatlodge} = C_w * \frac{Volume_{water}}{Volume_{airinlodge}} * V_{sw} * EF * ED * DCF * \frac{10^5}{27}$$

[36] Three liters/day of water ingestion are considered a reasonable upper bound intake amount for arid climates. Further support for this value can be obtained from reviewing the supporting literature used in the EPA *Exposure Factors Handbook* [U.S. EPA, 1997]. Briefly, a weighted average is obtained by assuming that increased water consumption of approximately 4 l/d occurs during the hot months (about one-third of the year), and a reasonable upper bound value of 2.3 l/d occurs during the remainder of the year.

[37] For this analysis, both the adult and child dose estimates are calculated using ICRP 60 methodology. Due to the inherent delays in the regulatory process, ICRP 60 methodology has yet to gain acceptance within the United States. However, child dose conversion factors are only available using ICRP 60 methodology. The adult dose estimates are provided using the same methodology (ICRP 60) as the child, for consistency.

Where:

- $Dose_{sweat\ lodge}$ = Committed effective dose from sweat lodge respiration (mrem/year)
- C_w = Contaminant groundwater concentration (pCi/l)
- $Volume_{water}$ = Quantity of water used in the sweat lodge (liters)
- $Volume_{air\ in\ lodge}$ = Air volume of the sweat lodge (m^3)
- V_{sw} = Breathing rate while in the sweat lodge (m^3/day)
- EF = Exposure Frequency (days per year exposed)
- ED = Exposure Duration (fraction of day exposed)
- DCF = Dose conversion factor (Sv/Bq)
- $10^5/27$ = Conversion factor from Sv to mrem and pCi to Bq

4.2.3 Groundwater Ingestion while Showering

An individual in either scenario is assumed to ingest 0.01 liters/day of water while showering. The shower water ingestion is a small fraction of the total ingestion of water per day.[38]

4.2.4 Groundwater Inhalation while Showering

An individual in either the Native American or rural resident scenario is assumed to shower for 15 minutes every day. Given the normal temperatures of a shower, about 0.1% of the total water volume is assumed to volatilize, with a corresponding amount of contaminants entrained in the volatilized particles. Other assumptions for calculating the dose include the breathing rate while showering and the total volume of the shower area. Given these parameters and assumptions, it can be shown that groundwater contaminants that are assumed to remain airborne will contribute a fraction of a mrem/y to an individual.[39] As the predicted impacts from any of the five groundwater contaminants are too small to warrant consideration in the alternatives, further estimates of groundwater inhalation while showering are not considered.

4.2.5 Dermal Absorption of Groundwater

Dermal absorption of radionuclides is not considered in this report. Unlike some chemicals, radionuclides are generally absorbed into the body very poorly [Yu, et al, 1993]. Tritium is an exception to this rule. Tritium, however, is found in very low concentrations in the groundwater, due to the short half-life and relatively small source term.

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[38] Potential exposure via inhalation while showering is generally only considered for volatile organic compounds [Yu, et al, 1993; U.S. DOE, 1996].

[39] For example, assuming a concentration of 500 pCi/l of Tc-99 in the water, 1 m^3/hr breathing rate, 0.1% volatilization for hot water 2.5 m^3 shower volume, 10-minute shower time (80 liters of water) for 365 days/year, and a dose conversion factor of 1.5×10^{-5} mrem/pCi, results in an estimated dose of 1×10^{-2} mrem/y.

4.3 Soil

Surface soil is contaminated through three mechanisms:

- The use of contaminated irrigation water
- The uncovering the contaminated waste through intruder activities such as digging a well
- The resuspension and redistribution of contaminated soil

The possibility for plants or animals to uncover or remove contaminated soil is discussed in Section 4.3.5. There are four methods by which exposure to contaminated soil can occur:

- Inadvertent ingestion (Section 4.3.1)
- Resuspension and inhalation (Section 4.3.2)
- External exposure (Section 4.3.3)
- Dermal exposure (Section 4.3.4)

In calculating the dose as a result of soil contamination, it is important to realize that soil contamination can occur through any combination of the three mechanisms. For example, an individual may live and grow crops outside of the contaminated area. Using irrigation water, he/she contaminates the soil over time as a result of the water being contaminated. If an intruder were present onsite, some additional, albeit small, contribution from resuspended material driven offsite could also contaminate the same soil. Similarly, for the intruder, soil would be contaminated through the use of irrigation water as well as through digging up contaminated waste and distributing it throughout the surface soil. For continuity, the calculation of the concentration of a contaminant in the soil is included in Section 3.5, as the equations for the soil concentration are linked with the food ingestion calculations.

4.3.1 Inadvertent Soil Ingestion

Ingestion of contaminated soil is possible as a result of transfer to vegetables, fruits, and hands [Kennedy and Strenge, 1992]. Although the amount ingested depends upon the activities performed and personal habits, a single conservative value is assumed. For the rural resident, 50 mg/day is assumed, while the Native American is assumed to ingest 200 mg/day. Children are also assumed to ingest 200 mg/day. The equation for calculating the ingestion dose is as follows [Kennedy and Strenge, 1992]:

Equation 4

$$Dose_{soiling} = C_{soil} * IR * ED * DCF * 100{,}000$$

Where:

- $Dose_{soiling}$ = Committed effective dose from the ingestion of soil

- C_{soil} = Concentration of soil (Bq/g)
- IR = Ingestion rate of soil (g/day)
- ED = Exposure duration (d/year)
- DCF = Committed effective dose conversion factor for ingestion (Sv/Bq)
- 100,000 = Conversion from Sv to mrem

A modifying factor may also be added to this equation to account for time spent outside of a contaminated area.

4.3.2 Soil Resuspension and Inhalation

Contaminated soil may also result in exposure due to resuspension and subsequent inhalation. For the intruder, exposure may occur from soil contaminated through irrigation water or through the uncovering of contaminated soil. For the offsite individuals, exposure from this pathway may occur from soil contaminated via irrigation water or from material dispersed from onsite. Note, however, for exposure to occur from contaminated material driven offsite, an intruder would have to gain access to the waste. Otherwise, the offsite soil is contaminated only with the radionuclides found in the groundwater.[40]

The resuspension factor does depend upon the activities that are being performed by the intruder. The highest dust loading is related to gardening activities, while the lowest is equated to time spent indoors. The equation for calculating the committed effective dose from inhalation is as follows [Kennedy and Strenge, 1992]:

Equation 5

$$Dose_{inhalation} = [(V_g * t_g * CDG * C * DCF) + (V_x * t_x * CDO * C * DCF) + (V_r * t_i * (CDI + P_d * RF_r) * C * DCF)] * 10^5$$

Where:

- V_g = Breathing rate for time spent in the garden (m^3/h)
- t_g = Time spent in the garden during a year (hours)
- CDG = Dust loading for activities taking place in the garden area (g/m^3)
- DCF = Inhalation committed effective dose, nuclide and age specific (Sv/Bq)
- V_x = Breathing rate for time spent outdoors (not in garden)(m^3/h)
- t_x = Time spent outdoors (not in garden) during a year (hours)
- CDO = Dust loading for outdoor (not in garden) activities (g/m^3)

[40] Offsite soil contamination from onsite activities can contribute through a number of pathways. The following calculations are therefore calculated as a percentage of the onsite dose. The integral of a time-dependent resuspension factor is 1.4×10^{-4} (d/m) [Anspaugh, 1998]. By multiplying the air resuspension integrated over a year by the deposition velocity (0.001 m/s), by the 0.176 fraction of time the wind blows toward the offsite MEI direction, and by 86,400 s/day, the product yields a dimensionless factor by which the onsite dose from various pathways can then be multiplied. Offsite ingestion and external doses will not exceed 0.2 % of the onsite doses.

- V_r = Breathing rate for time spent indoors (m^3/h)
- t_i = Time spent indoors during a year (hours)
- CDI = Dust loading for indoor activities (g/m^3)
- P_d = Indoor dust loading on floors (g/m^2)
- RF_r = Indoor resuspension factor (per meter)
- 100,000 = Conversion from Sv to mrem

The indoor portion of the above equation differs slightly from the outdoor portion, as it includes contributions from materials blown and soil tracked into the house and resuspended [Kennedy and Strenge, 1992].

4.3.2.1 Calculation of the Offsite Dose Due to Resuspension from Onsite

Section 4.3.2 provides a discussion and method for determining the relative impact to offsite locations as a result of onsite contamination. This method calculated the impact as a result of accumulated soil contamination over time. Soil inhalation, however, depends upon the contaminant concentration in the air, and is determined somewhat differently. The offsite air concentration at any given time would be significantly less than the corresponding accumulated deposition that results in the 0.2% of dose factor calculated in the footnote supporting Section 4.3.2. However, for calculational ease, it is assumed that the contribution to inhalation dose from onsite resuspended material is 0.2% of dose as well.

4.3.3 External Exposure to Soil

External exposure to contaminated soil is generally only a potential hazard for intruder activities.[41] Offsite exposures only occur from the groundwater contaminants, which are not external exposure hazards, or from materials driven offsite (from onsite), which would be low in concentration (<0.2% of the onsite dose). For the intruder, the possible contaminants include the entire waste inventory.

In order for an intruder to bring the contaminated material to the surface onsite, a 12-inch (30 cm) diameter well is assumed to be drilled (see the intruder construction scenario) to 360 feet (110 meters) (50 feet past the presumed groundwater table). Of that 360 feet of material, 37 feet (11.3 meters) are assumed to be contaminated with a homogeneous mix of the source material from the low-level waste.[42] This contaminated material is uniformly spread over a 16,000 square foot area (1,500 square meters) [U.S. NRC, 1981, Napier, et al, 1984]. The depth of the contamination is six inches (15 cm),

[41] As discussed in the inadvertent soil ingestion section, groundwater contaminants are not gamma emitters and would not pose an external hazard. The resuspended material from onsite deposited offsite is at most 0.2% of the onsite dose. External contributions from all materials are considered in the supporting documentation to this analysis.

[42] Recent trenches have a depth of 45 feet, 37 of which are dedicated to low-level waste. The remaining 8 feet are clean fill to grade.

as the material is assumed to be uniformly tilled.[43] The 1,500 square meters allow the calculations to approximate an infinite plane [Napier, et al, 1984] for external dose calculations.

In order to accurately calculate the ingrowth of the progeny (for the intruder) and perform further external exposure calculations, the computer code MICROSHIELD [Grove Engineering, 1998] is used. The MICROSHIELD code calculates the parent and progeny concentrations as well as an estimate of the effective dose equivalent, using ICRP 51 methodology [ICRP 51, 1987].

The external dose contribution analysis for both indoor and outdoor scenarios is performed in the following manner:

1. The concentration in the waste volume was estimated by taking the total source activity per radionuclide and dividing it by the total mass of waste and other fill in the active waste region.[44] The estimate excludes the mass of soil between trenches at the depth of the waste.

2. The volume of waste (0.8 cubic meters) is then removed and uniformly spread over the top 15 centimeters of soil to an area of 1,500 square meters.

3. This surface concentration is entered into the MICROSHIELD code in the form of a perfect disk source, with the dose point (the individual) in the center. The soil used for the analysis is a Nevada Test Site (NTS) dry, sandy soil [Eckerman and Ryman, 1993]. The NTS soil is sufficiently close to the cover material that will be used at the LLRW disposal site.[45]

4. MICROSHIELD calculates the estimated contribution to dose, using the appropriate buildup and attenuation factors for the soil and air [Grove Engineering, 1988]. As a check on results, the concentrations obtained from the output of the MICROSHIELD code are also used as the input for analysis using Federal Guidance Report (FGR) #12 [Eckerman and Ryman, 1993]. The tables for uniform contamination to 15 centimeters were used. These tables are based upon an infinite plane source.

The general formula used for calculating the external effective dose equivalent for outdoor exposure is as follows:

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[43] A volume of 0.8 cubic meter of contaminated material is removed from the well. The 15-cm mixing provides a realistic depth of soil for farming use and also serves to maximize the potential impacts of uptake to plants.

[44] The volume used for dilution has been modified from the 50-million cubic feet value used by US Ecology. DOH instead used the volume of the waste area excluding the cover material. In order to calculate this, DOH determined the fill efficiency for each trench (amount of waste per total waste area). This information was then used to determine the total waste area volume for the year 2056, by dividing the projected waste inventory of 20 million cubic feet by the fill efficiency [Ahmad, 1988].

[45] This soil also has the added benefit of being analyzed for comparison with the results of Federal Guidance Report #12 [Eckerman and Ryman, 1993].

Equation 6

$$ExternalDose = C * DCF * ED * 3600 * \frac{1500}{2500}$$

Where:

- External dose = Dose in Sieverts (multiply by 10,000 to obtain dose in mrem)
- C = Concentration ($Bq*m^{-3}$)
- DCF = Dose conversion factor, nuclide specific ($Sv*s^{-1}*Bq^{-1}*m^3$)
- ED = Exposure duration (hours/year)
- 3600 = Conversion from hours to seconds
- 1500/2500 = Corrects for the time spent within the contaminated area

In the child analysis, the values of ED and time spent within the contaminated area are modified to account for attending an offsite school.

As the contribution is from an external field, a whole body dose is assumed and can be added to the effective dose calculated from internally deposited material. For calculational ease, a shape factor[46] of one (1) was assumed for time spent within the 1,500 square meter contaminated area. Time spent outside the 1,500 square meter area was considered to have a shape factor of zero, thereby contributing nothing to the calculated dose. This assumption is conservative, as the time spent within the 1,500 square meter area would rarely be a perfect geometry, and time spent near the edge would be about half.

Perhaps the largest unknown is the estimated time that an individual spends outside. For the rural resident intruder, since the assumption is made that the individual lives and grows some food at the LLRW disposal site, it is assumed that 60% of his time is spent indoors [Yu, et al, 1993], and 40% outdoors.[47] The Native American intruder is assumed to spend equal amounts of time both indoors and out.

The external radiation contribution from time spent indoors is calculated in a similar manner to the calculation for the time spent outdoors. It is assumed that contamination is not directly underneath the foundation of the house.[48] An indoor shielding factor of 0.33 [Kennedy and Strenge, 1992] is utilized to account for the shielding provided by the structure of the home, the reduction from an infinite plane source as the home is at the boundary of the contaminated area, and a further reduction to account for time spent indoors away from the walls. The exposure time indoors is 60%, or 5,250 hours

[46] The shape factor is a correction that takes into account irregularly shaped contaminated areas.
[47] The indoor time estimates for this analysis are somewhat lower than the estimates provided in a review performed by the U.S. EPA [U.S. EPA, 1992]. The lesser amount of time spent indoors as compared to the estimated United States average is expected to result from the greater amount of food grown individually.
[48] Directly underneath means contaminated waste from the well cuttings, not the contaminated waste still buried in the trenches.

per year for the rural resident intruder, and 4,380 hours per year for the Native American intruder. The formula for indoor exposure is:

Equation 7

$$ExternalDose = C * DCF * ED * 3600 * 0.33$$

Where:

- 0.33 = Indoors shielding factor[49]

4.3.4 Dermal Exposure

The absorption fraction for radionuclides on the skin that are absorbed into the blood is generally small, and with the exception of H-3, is not further considered in this analysis. Chemical dermal contact of volatile organics, by comparison, has significantly higher absorption rates and has the potential for contributing to exposure.

In addition to skin absorption, dermal contact with radionuclides may also pose a risk, assuming the contaminant is of a sufficient concentration. Generally speaking, for a contaminant on the skin to pose a hazard, the radionuclide must be a strong beta or gamma emitter. In these instances, the risk from exposure does not sufficiently contribute to dose, as the contamination is on the arms and legs. The hazard from these exposures is from burns or ulceration, assuming the contamination is present long enough or in sufficient concentration. As an example, the strongest external hazard present in post-closure analysis is Cs-137. An assumption of closure in the year 2056, with potential access in 2163, results in a Cs-137 concentration of 11 pCi/g to the intruder. To calculate the concentration per centimeter on the body would be as follows:

Equation 8

$$SkinContamination = C_s * SAF$$

Where:

- C_S = Soil contamination in pCi/g
- SAF = Skin adherence factor (g/cm^2)

A standard skin adherence factor is 0.2 mg/cm^2 [U.S. DOE, 1996]. For cesium, the result is a concentration of 2.2x10^{-3} pCi/cm^2. This contaminant concentration would need to be at least nine (9) orders of magnitude greater before deterministic risks such

[49] Without considering the shielding provided by the housing structure, the MICROSHIELD code estimates that the external dose rate would be reduced by approximately 90% for an individual standing 10 feet from the edge of the contaminated area (the wall of the home). The indoor shielding factor of 0.33 is therefore considered conservative.

as skin burns became an issue.[50] Dermal exposure for radionuclides is therefore not included in this analysis.

4.3.5 Direct Contact with Buried Waste

Potential biotic intrusion (i.e., plant roots and burrowing animals) into the waste trenches was evaluated. The proposed depth of the trench cover varies from a minimum of 11'6" for the Site Soils Cover - , to 16'4" for the Proposed Action and Enhanced closure alternatives. In addition, three of five closure alternatives include covers with characteristics that inhibit penetration by plant roots (e.g., bentonite layer, asphalt). U.S. DOE (U.S. DOE, 1995) summarized the published information on plant rooting and animal burrowing depths for Hanford, that included a study by Klepper on the rooting depths of deep-rooted plants common to the 200 Areas that are adjacent to the LLRW disposal site. The deepest burrowing animal was the harvest ant at 8.9 feet, and the badger was the deepest burrowing mammal at 8.2 feet [U.S. DOE, 1995]. Klepper found that eight of the 14 plant species investigated had average maximum rooting depths exceeding 4.9 feet. The species with the greatest average maximum rooting depth are antelope bitterbrush (9.7 feet), big sagebrush (6.6 feet), and spiny hopsage (6.4 feet). Variability in maximum rooting depth among individual plants of a species was low (i.e., coefficient of variation ranged from 0.03 to 0.20 among species), suggesting that rooting depth may be limited by available soil moisture. Furthermore, the ecological risk assessment regulations currently under development by the Department of Ecology state that a terrestrial evaluation can be completed and no further analysis required for sites where the soil contamination is at least six feet below the soil surface. Based upon this information, the direct contact exposure pathway of plants or animals to waste buried under covers will not be considered for all the closure alternatives.[51]

4.4 Air

This section describes the process for evaluating the expected dose from exposure to gaseous radionuclides at the LLRW disposal site. This analysis considers three potential contributors to dose: radon (and progeny), carbon 14, and tritium. Chlorine 36 is also a potential gaseous emitter but is considered to impact via the groundwater. The discussion for the three radionuclides describes the numerous considerations involved in analyzing the potential impact to individuals indoors, outdoors, and offsite.

Of potential concern is the possible impact to LLRW disposal site boundary locations prior to the end of institutional control. Due to the long half-life of radium 226 (the parent of radon) and of carbon 14, the offsite estimates for these two radionuclides can be applied to any time period during the institutional control period, due to the small amount of decay. Tritium, due to its short half-life, decays considerably during the

[50] Based upon the NCRP-recommended limit of 75μCi-hrs of exposure [NCRP, 1989].
[51] This entire chapter is borrowed from the *Chemical Risk Assessment for the Commercial Low-Level Radioactive Waste Disposal Facility, Richland, Washington* [Kirner Consulting, Inc., 1999].

institutional control period. Specific calculations are therefore performed for tritium to estimate the potential impact at the proposed LLRW disposal site closure date.

4.4.1 Radon Contribution Analysis

Radium 226, with a half-life of 1600 years, alpha decays to radon 222 with a half-life of 3.8 days. Radon is a gas, and as such, a fraction of the radium 226 that decays escapes the confines of the soil column and migrates toward the surface. This diffuse radon can accumulate in houses through cracks in the floor, around floor penetrations (such as drainpipes), and through the concrete floor. A portion of the radon in the air is respirated and retained in the lung where the radon daughters (Po-218, Bi-214, Pb-214, and Po-214) deliver a dose that is approximately 100 times greater than the dose of radon 222.[52]

For the proposed alternatives, cover depth and the addition of a clay layer are two controllable factors that drive the estimated radon flux from the soil. When considering the thickness of the cover for radon reduction potential, gravel layers are not assumed to have any mitigating effect. Clay, however, has a tremendous impact on radon emanation. A clay barrier is estimated to reduce the predicted emanation rate by a factor of 2.5. Enhanced barriers such an asphalt or a geomembrane are essentially impermeable while intact.

The radon discussion is divided into three sections: indoor radon, outdoor radon to the intruder, and offsite radon contribution. Radon is predominately a contributor to dose while indoors, as the gas has a greater opportunity to accumulate in a home without the benefit of the free exchange of air. As a result, a majority of the focus is spent on determining the largest contribution to dose: the indoor radon pathway.

4.4.1.1 Indoor Radon Contribution

One driving assumption for the indoor radon dose is that an intruder will build a basement whose depth does not exceed the seven-foot depth of the barriers (the sand/bentonite layer) found in most of the designed covers, thus reducing the dose received from the radon daughters by a factor of about 2.5. Building requirements for access and egress from a basement dictate that a seven-foot excavation depth is reasonable for new construction homes [Aleshire, 1997]. Based upon this information, DOH assumed a seven-foot building foundation excavation depth.

4.4.1.1.a Methodology

[52] In addition, Rn-220 (thoron), the daughter of Th-232, was evaluated as not being capable of significantly contributing to dose, as the half-life for Rn-220 is sufficiently short that diffusion through the cover layer is not considered possible, due to the significant decay of the Rn-220 concentration with depth [NCRP, 1987a]. For Th-232 removed by intruder activity to the surface soil, the inhaled dose from thoron is about one seventh that of radon [NCRP, 1987a], assuming equivalent concentrations of Rn-222 (radon) and Rn-220 (thoron).

The conversion of a radium soil concentration to a dose to an individual involves a number of assumptions and approximations. The flow path of working from a soil concentration to a dose using deterministic values is discussed below.

For modeling purposes, the following assumptions were used:

- The cover layers beneath the basement slab were assumed to be a single barrier (if present), followed by a layer of site sand.
- The characteristics of the site sand are assumed to apply uniformly to the cover. This is an inherently conservative assumption in that all covers (with the exception of the site soils cover) include a thick vegetative layer that would have a significantly greater moisture retention fraction (and greater radon attenuation capability) as compared to a similar layer of site soils).[53]
- The waste volume was assumed to be approximately 35 feet deep. The radon flux from the waste volume was calculated using the formulas provided in NRC Regulatory Guide 3.64 [U.S. NRC, 1989]. Further details regarding the flux calculations are located in the supporting documentation [Thatcher, et al, 1998].
- The waste for radium is segregated into a number of depths to accurately account for the depth below grade of the waste disposed over the years. The four depths for waste used are 3 feet, 8 feet, 16.5 feet, and greater than 23 feet [Elsen. 2003].
- Future radium waste is split between an assumed breakout of 10% class A waste and 90% Class c waste. Future Class A waste is assumed to be buried at 8 feet below grade whereas the Class C waste is assumed to be buried at greater than 23 feet below grade. 4.2 Ci/y of radium 226 is assumed to be accepted each year for all future waste.
- The performance of all barriers (i.e. bentonite, asphalt, and gcl/geomembrane layers) is assumed to degrade over time. The degradation is assumed to take the form of an increased porosity as a result of settlement of the waste.
- The enhanced asphalt and gcl/geomembrane covers are assumed to completely impede radon emanation over the first 500 years.

- The formula for the diffusion coefficient is based upon updated information [Rogers and Nielson, 1991]. The formula is as follows:

Equation 9

$$D_c = D_o * p * e^{(-6*S*p-6*S^{14}*p)}$$

Where:

- D_c = Diffusion coefficient for radon in soil (cm^2/s)
- D_o = Diffusion coefficient for radon in air (cm^2/s)
- p = Soil porosity

[53] The vegetative cover has no impact on indoor dose calculations, as this layer is assumed to be removed when the foundation for the home is built.

- S = Volume fraction of water saturation[54]

  This updated diffusion coefficient equation is based upon over 1,000 additional radon diffusion coefficient measurements for soils, and over 600 additional measurements for uranium mill tailings than is recommended in NRC Reg. Guide 3.64. The updated empirical equation generally results in lower estimates of the diffusion coefficient, as compared with the previous equation.

- DOH modified the source term provided in the US Ecology closure plan, to account for a portion of the radium disposed in a sealed container.[55] The reduction in the radon diffusion coefficient was accounted for by reviewing the disposal records for 1987, 1988, 1989 [U.S. NRC, 1990], 1994, 1995 [Blacklaw, 1996], and 1996 [Elsen, 1997]. The discrete (sealed) radium concentration is 81% of the total radium disposed for those years. The NRC [U.S. NRC, 1982] requires the assumption that all material (i.e., concrete) will degrade within 500 years. As a result, at 500 years following closure, the entire radium activity is considered available for diffusion.

- A conservative 20% reduction factor [Landman and Cohen, 1983] is applied to the radon flux value to take into account the decreased emanation rate through a cracked concrete floor (concrete without cracks would have an emanation rate of less than 1%, as compared to the bare soil flux).[56]

- Assuming a ventilation rate of 0.5 hr^{-1} [Yu, et al, 1993], the calculated steady-state radon concentration is calculated. This concentration includes a factor [Marcinowski, et al, 1994] to correct basement concentrations to concentrations in living spaces.[57] The formula for calculating the indoor concentration is as follows [Yu, et al, 1993]:

  Equation 10

$$C_i = \frac{(\frac{J_i}{H} + v * C_o) * 0.38 * 0.20}{(\lambda + v) * 1000}$$

  Where:

- C_i = Indoor concentration (pCi/l)

[54] Also called the moisture saturation fraction in the RAETRAD code. This tracks the moisture carrying capacity of the soil, not how much moisture is in the soil at any given time.
[55] The radium disposed as a sealed source is generally contained within 2500 psi concrete and would not contribute to the overall radon gas emanation rate.
[56] The relatively large fraction of radon passing through the cracked concrete floor also serves to model for pressure-driven radon entry (advection), in addition to diffusion.
[57] The National Residential Radon Survey conducted in 1989 and 1990 collected data for all spaces of a home. Total basement concentration (living and non-living spaces) was 122.1 Bq/m^3 (arithmetic mean). The average concentration in a home was found to be 46.3 Bq/m^3. The resulting correction from basement to total home is 0.38.

- C_o= Outdoor concentration (pCi/l)
- J_i= Radon flux (pCi/m^2*s)
- H = Room height (m)
- v = Ventilation rate (s^{-1})
- λ = Decay constant of radon (s^{-1})
- 1000 = Conversion from m^3 to liters
- 0.38 = Corrects basement reading to predominate level of living space
- 0.20 = Provides an adjusted bare floor diffusion rate to take into account a cracked concrete floor

- The concentration of radon daughters (the contributors to dose) in the air (of a room) is significantly less than the concentration of radon itself, due to a number of factors. Those factors include radioactive decay, plateout (settling onto walls and other surfaces of a room), and physical removal by ventilation. The application of an equilibrium correction factor 'F' accounts for the lower concentration of radon daughters measured in an environment. The equilibrium F factor is highly correlated with ventilation rates in a home [Swedjmark, 1983]. As ventilation rates for United States homes range from .35 to 1.5 exchange volumes per hour [Yu, et al, 1993], the equilibrium equivalent concentration (EEC)[58] is approximately 33% to 50% [Swedjmark, 1983] of the radon concentration.[59]

- The equilibrium concentration of radon daughters in a home is then converted to a working level[60] (EEC/100), a common term for expressing radon exposure. The formula for calculating the working level (WL) is:

  Equation 11

$$WL\ (pCi/l) = 0.00104[^{218}Po] + 0.00514[^{214}Pb] + 0.00382\ [^{214}Bi]$$

- The result is then converted to working level months per year (WLM/year). The WLM/year is the exposure rate in WL, multiplied by the hours of exposure (per year for residential exposures), divided by 170 hours (the number of hours per month that a uranium miner typically spends in the mines). The onsite rural resident is assumed to spend 60% of his/her time indoors, resulting in an exposure time of approximately 5,250 hours/year. The formula for the WLM/year is as follows:

  Equation 12

$$WLM/yr = \frac{WL * ExposureTime}{170 hours}$$

[58] EEC is the radon concentration in equilibrium with the short-lived daughters.
[59] NOTE: A linear equation for the radon concentration as a function of ventilation rate was used, as the NCRP-recommended value (.5/.3/.2) for Po-218, Bi-214, and Pb-214 does not account for fluctuations in the ventilation rate.
[60] Working level is defined as any combination of short-lived radon daughters in one liter of air that will result in the ultimate emission of 1.3 X 10^5 MeV of potential alpha energy [NCRP, 1988].

- The effective dose to an individual is estimated by using an effective dose per unit exposure conversion factor of 830 mrem/WLM [Porstendorfer and Reineking, 1999]. This value is based upon ICRP 66 [ICRP, 1994] lung dosimetry, and estimates of 'normal' indoor particle concentrations.[61]

4.4.1.2 Outdoor Radon Contribution

For the intruder scenario, the individual also receives a dose from the ambient concentration of radon while outdoors. Two sources of radon contamination exist for the intruder; the first is the buried contaminated waste on which the intruder lives, and the second is the contaminated material brought to the surface as a result of drilling a well. The combination of these two sources is added to provide the estimate of the outdoor radon contribution.

The surface flux estimate can then be utilized to determine an ambient air concentration onsite, using the following formula [Yu, et al, 1993]:

Equation 13

$$C_{Radoninair} = \frac{\{0.5 * EVSN * \sqrt{A}\}}{\{H_{mix} * U\}}$$

Where:

- $C_{Radon\ in\ air}$ = Average concentration of radon in air over a contaminated area (pCi/m^3)
- 0.5 = Default time fraction wind is blowing toward individual (dimensionless)
- EVSN = radon flux (pCi/m^2s)
- A = Area of contaminated zone (228,000 m^2)
- H_{mix} = Height of interest for uniform mixing (1 m for plants, 2 m for adults)
- U = Average wind speed (3.4 m/s) [Gleckler, et al, 1995]

4.4.1.3 Offsite Radon Contribution

Contributions to a resident at the LLRW disposal site boundary can only occur via gaseous diffusion of radon emanating onsite. The gaseous concentration offsite is determined by using the onsite surface flux estimate, which varies depending upon the cover material and layers. The flux is then multiplied by the area of the assumed contamination. For the gaseous emitters, this is the 228,000-square meter area of the

[61] Although dosimetry is used in this EIS to estimate the resulting dose, the ICRP has concluded that the use of epidemiology of radon in mines is more direct, and involves less uncertainty. It is therefore more appropriate to use the ICRP 65 report than the indirect use of the epidemiology of low-LET radiation from Japanese data [ICRP 65]. The ICRP recommends that the dosimetric model should not be used for the assessment and control of radon exposures. Nevertheless, Porstendorfer's estimates appear to be reasonably close to the estimates from BEIR VI, but more conservative than the ICRP 65 recommendations, by about a factor of two.

LLRW disposal site. This provides a total LLRW disposal site release rate. This value is then multiplied by the dispersion coefficient for a contaminant at a specific offsite distance [US Ecology, 1996]. The maximum offsite distance is east-southeast of the LLRW disposal site. The estimates are calculated for the maximum predicted location. The formula for the calculation is as follows:

Equation 14

$$C_a = RadonFlux * Area_{site} * \frac{\chi}{Q} * \frac{1}{1{,}000}$$

Where:

- C_a = Air concentration offsite (pCi/l)
- Radon Flux = ground level emission rate (pCi/m^2*s)
- $Area_{site}$ = Area of trenches (m^2)
- X = The offsite air concentration at the location of interest (pCi/m^3)
- Q = product of the radon flux and the LLRW disposal site area (pCi/s)

1/1,000 = converts air concentration from m^3 to liters

4.4.2 Carbon 14

Carbon 14 is modeled separately from other radionuclides, due to the ever-present nature of carbon in the environment. Carbon 14 presents only an internal risk to humans, as the energy of the beta particle is too low to cause a concern for external exposure. For the carbon 14 modeling, it is assumed that equilibrium exists between the soil, plants, and humans. Carbon 14 is modeled with equal fractions being released as a gas and through the groundwater[62]. The methodology for the incorporation of carbon 14 via the air and water pathways is included below.

One of the major difficulties in estimating the dose from carbon 14 is determining the portion of the source volume that is available for biodegradation. Once the source term has been established, the carbon 14 flux emanating through the cover must be estimated. Dr. Man-Sung Yim calculated these initial portions of the dose calculation at North Carolina State University [Yim, 1997]. To summarize Dr. Yim's report:

- Approximately 55% of the total carbon 14 inventory is assumed to be biodegradable
- For the air pathway, the predicted surface flux at the end of the institutional control period is 6.4x10^{-6} Ci/m^{2}y for the realistic estimate, and 10.7x10^{-6} Ci/m^{2}y for the conservative case.[63]
- The difference between the two flux estimates results from the assumption that all of the organic materials are assumed to be biodegradable, regardless of chemical form

[62] Rood, A., *Groundwater Concentrations and Drinking Water Doses with Uncertainty for the U.S. Ecology Low-Level Radioactive Waste Disposal Facility*, Richland Washington, March 2003.

[63] These estimates are corrected for the upward revision of the source term from the 3670 curies used in the original calculations, to the 5,247 curies used in the final calculations. The 5,247 curies accounts for the projected disposal of C-14 through the year 2056.

(conservative case), whereas the expected chemical form of carbon 14 in various waste streams is taken into account for the biodegradability estimation in the realistic case.

The surface flux estimate can then be utilized to determine an ambient air concentration using the following formula [Yu, et al, 1993]:

Equation 15

$$C_{C\text{-}14inair} = \frac{\{0.5 * EVSN * \sqrt{A}\}}{\{H_{mix} * U\}}$$

Where:

- $C_{C\text{-}14\ in\ air}$ = Average concentration of carbon 14 in air over a contaminated area (pCi/m^3)
- 0.5 = Default time fraction wind is blowing toward individual (dimensionless)
- EVSN = Carbon 14 flux (pCi/m^2s)[64] [Yim, 1997]
- A = Area of contaminated zone (228,000 m^2)
- H_{mix} = Height of interest for uniform mixing (1 m for plants, 2 m for adults)
- U = Average wind speed (3.4 m/s) [Gleckler, et al, 1995]

The flux estimate is a total carbon 14 flux per year; however, a portion of this carbon 14 is in the form of methane (CH_4) and unavailable for photosynthesis. The fraction of the carbon 14 that is methane is assumed to be 50%[65] [Tchobanoglous, et al, 1993].

The next step is to calculate the concentration in plants due to the concentration in air and soil [Yu, 1993].

Equation 16

$$C_{C\text{-}14,v} = C_{c,v} * \left\{\left\{F_a * \frac{C_{C\text{-}14,a}}{C_{C,a}}\right\} + \left\{F_s * \frac{S_{C\text{-}14}}{S_C}\right\}\right\}$$

Where:

- $C_{C\text{-}14,v}$ = Concentration of carbon 14 in plants (pCi/kg)
- $C_{C,v}$ = Fraction of stable carbon in plants[66] (0.1)

[64] The flux is based upon a homogenous carbon 14 source term. The realistic flux estimate is used for this analysis and is itself conservative, due to the assumptions made in determining the biodegradable portion.

[65] Low-level radioactive waste landfills have been shown to be chemically similar to sanitary landfills [Husain, et al, 1979]. Although the rate of production of gases is small when compared to sanitary landfills [Kunz, 1982], the composition of the gases, over time, is expected to be similar to sanitary landfills.

[66] Take the carbon in vegetation of 0.45 kg C/kg dry [Napier, et al, 1988] and multiply it by the dry-to-wet weight conversion factors [Kennedy and Strenge, 1992] (0.18, 0.25, and 0.20 for fruit, other vegetables,

- F_a = Fraction of carbon in plants derived from carbon in air (0.98) [Yu, et al, 1993]
- F_s = Fraction of carbon in plants derived from carbon in soil (0.02) [Yu, et al, 1993]
- $C_{C,a}$ = Concentration of stable carbon in air (1.6×10^{-4} kg/m^3) [Yu, et al, 1993]
- $S_{C\text{-}14}$ = Concentration of carbon 14 in soil (pCi/kg)
- S_C = Fraction of soil that is stable carbon (0.03) [Yu, et al, 1993]

The contaminated zone where the material is buried is located approximately five meters beneath the surface for all closure alternatives, with the exception of the Site Soils alternative. Soil to plant uptake can occur through the irrigation of plants and the subsequent contamination of the upper soil column. This water pathway, however, is assumed to be a very small part (2%) of the overall plant concentration of carbon 14 [Yu, et al, 1993]. The majority of plant contamination (98%) is due to intake of carbon during photosynthesis. As the flux is assumed constant over time, this plant concentration is an assumed equilibrium value.

The final step in the estimate of the dose contribution to an onsite individual is to calculate the total carbon 14 intake on an annual basis. Using the NRC-recommended consumption values for the general population [Kennedy and Strenge, 1992] and the EPA estimates for locally grown products [U.S. EPA, 1991], the estimated consumption of fruit consumption is 13.8 kg/year, of leafy vegetables is 4.4 kg/year,[67] and of other vegetables is 20.4 kg/year, from which a total intake of 38.6 kg/year is obtained. This results in a combined annual carbon 14 intake of 3.8 kg per year, assuming that all consumed carbon is in the form of carbon 14.

For the Native American, using the recommended consumption values [Harris and Harper, 1997] and estimates of locally grown products, the estimated consumption of local fruit is 52.3 kg, of leafy vegetables is 40.2 kg, and of other vegetables is 37.4 kg. This results in a combined annual carbon 14 intake of 12 kg per year, assuming all consumed carbon is in the form of carbon 14.

Using the dose conversion factor of 5.64×10^{-10} Sv/Bq [Eckerman, et al, 1988], the resulting formula to estimate the dose is:

Equation 17

$$Dose(mrem/y) = \frac{C_{C14,v}}{g} * \frac{carbonIntake}{y} * \frac{Bq}{27pCi} * \frac{5.64E-10Sv}{Bq} * \frac{E+05mrem}{Sv}$$

Individuals residing within the area in which the carbon 14 flux is emanating will also receive a dose contribution as a result of inhalation. However, due to the low air

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and leafy vegetables, respectively), weighted by the respective consumption of homegrown produce recommended by the EPA [U.S. EPA, 1991].

[67] The EPA does not provide a separate value for the intake of leafy vegetables. The leafy vegetable consumption rate is therefore calculated using the ratio of the leafy vegetable fraction recommended by Kennedy [Kennedy and Strenge, 1992], multiplied by the consumption rate of vegetables recommended by the EPA.

concentration and an even lower dose conversion factor (6.2×10^{-12} Sv/Bq), the resulting dose contribution is approximately 180 times lower than the plant ingestion contribution.

4.4.2.1 Offsite Impact from Carbon 14

The calculations to the offsite individual from carbon 14 are performed exactly like the method provided for the onsite dose calculated above. The only parameter that changes is the carbon 14 flux estimate.

4.4.3 Tritium Analysis

Tritium analysis, similar to carbon 14 analysis, is performed separately from other radionuclides due to the ever-present nature of hydrogen in the environment. Tritium presents only an internal hazard, due to the extremely weak beta emission of the radionuclide.

Based upon the potential for offsite impact during the institutional control period, the modeling of the expected dose to an offsite individual at the maximum downwind location is calculated to determine the contribution from both contaminated groundwater as well as tritium gas escaping through the surface of the facility.[68] This modeling assumes that the source term is released both as a gas and corrects the groundwater release fraction to match the currently observed groundwater contamination beneath the LLRW. The methodology for the incorporation of carbon 14 via the air and water pathways is included below.

4.4.3.1 Tritium Contributions Via the Air Pathway

The tritium surface flux is estimated using the RADON computer code [U.S. NRC, 1989a]. For the 2056 closure date, the predicted surface flux is 0.5 pCi/m²s. Using the formula provided in Section 4.4.1.3, with a dispersion coefficient of 2.8×10^{-5} for a location 330m ESE (from the center of the LLRW disposal site), the estimated ambient concentration is 0.0029 pCi/l.

Similarly, the surface flux estimate can then be utilized to determine an ambient air concentration onsite, using the following formula [Yu, et al, 1993]:

Equation 18

$$C_{h-3inair} = \frac{\{0.5 * EVSN * \sqrt{A}\}}{\{H_{mix} * U\}}$$

Where:

[68] Rood, A., *Groundwater Concentrations and Drinking Water Doses with Uncertainty for the U.S. Ecology Low-Level Radioactive Waste Disposal Facility*, Richland Washington, March 2003.

- $C_{H\text{-}3\ in\ air}$ = Average concentration of carbon 14 in air over a contaminated area (pCi/m^3)
- 0.5 = Default time fraction wind is blowing toward individual (dimensionless)
- EVSN = Tritium flux (pCi/m^2*s) [Yim, 1997]
- A = Area of contaminated zone (228,000 m^2)
- H_{mix} = Height of interest for uniform mixing (1 m for plants, 2 m for adults)
- U = Average wind speed (3.4 m/s) [Gleckler, et al, 1995]

For example, using the year 2005 as the proposed closure date, with institutional control lapsing in the year 2112 (it will take seven years to close the LLRW disposal site), the estimated 1,100 curies of tritium remaining will result in a surface flux of 0.02 pCi/m^2*s, resulting in an onsite air concentration of 0.0011 pCi/l.

4.4.3.2 Tritium Contributions Via the Groundwater Pathway

The groundwater modeling for the site assumes that the tritium is released entirely as a liquid and not as a gas[69]. Likewise, the gaseous modeling assumed that 100% of the tritium source term escapes as a gas. The estimated tritium contributions should therefore be considered conservative.

4.4.3.3 Tritium Dosimetry

The NCRP developed a model for estimating the contributions from tritium by assuming or knowing concentrations in air, water, plants and animals [NCRP, 1979]. The NCRP dose factor for tritium at equilibrium is $9.5x10^{-5}$ mrem/year per pCi/L. In this instance, the NCRP model is utilized by assuming that the predicted groundwater concentrations are in equilibrium with the plants and animals and combined to the predicted downwind air concentration. The formula for estimating the contribution from all pathways is as follows:

$$\text{Dose} = \frac{D_i * C_w + 1.56 * C_f + 0.22 * C_a}{(D_i + 1.78)} * DCF$$

Where:
D_I = Drinking water intake rate (L/d), scenario specific value
C_w = Tritium concentration in drinking water (pCi/L)
C_f = Tritium concentration in foodstuffs (pCi/L)
C_a = Concentration in air (pCi/L)
1.56 = Assumed liquid intake from foodstuffs (L/d)
0.22 = Assumed liquid intake from skin absorption and inhalation (L/d)
DCF = Dose conversion factor ($9.5x10^{-5}$ mrem/year per pCi/L)

[69] Rood, A., *Groundwater Concentrations and Drinking Water Doses with Uncertainty for the U.S. Ecology Low-Level Radioactive Waste Disposal Facility*, Richland Washington, March 2003.

The equation provided above was slightly modified from that in the NCRP 62 to account for the greater drinking water intake rate. The tritium concentration in foodstuffs is assumed to be equal to the concentration in groundwater.

For the Native America scenarios, the additional contributions due to skin absorption and inhalation of tritium during sweat lodge use were also considered and were based upon the time use estimated in the Native American exposure scenarios. For skin absorption, the recommended uptake rate from Osborne [Osborne, 1972] of 10 μCi/min/μCi/L was used as the basis for estimating the absorption rate of tritium through the skin. The tritium concentration in groundwater was converted to an air concentration by assuming a vapor density of 0.2 L/m^3 and a breathing rate of 1.2 m^3/hr. The exposure times of 1 hr/day for the Native American Adult and 26 hr/year for the Native American Child were used with the overall dose estimated based upon the inhalation and ingestion dose conversion factor (DCF) of 1.8 E-11 Sv/Bq for tritium. The formula for calculating the tritium contribution from the sweat lodge is as follows:

$$\text{Dose} = C_a * (I_a + I_{br}) * DCF * 3703$$

Where:
I_a = Water intake via absorption (L/y)
I_{br} = Water intake via breathing (L/y)
DCF = 1.8 E-11 (Sv/bq)
3703 = Conversion from Sv/bq to mrem/yr

4.5 Food

Food contamination results from contamination in one or all of the three primary exposure routes: air, water, and soil. Food ingestion is included as its own pathway in order to clearly provide its impact on the predicted dose. The food analysis is divided into two categories: impacts that result from the ingestion of fruit and vegetables, and impacts that result from the ingestion of meat and dairy products.

4.5.1 Ingestion of Fruit and Vegetable Products

The analysis considers two mechanisms by which food contamination can occur: through irrigation, or through the uncovering of waste by the intruder. The analysis from the direct removal of waste and subsequent use for crops simplifies the analysis presented for estimating the impact from irrigation, as the soil concentration is at a maximum initially. Soil contaminated by irrigation must build up in concentration over time.

4.5.1.1 Ingestion of Fruit and Vegetable Products Contaminated by Overhead Irrigation Spray

The calculation of the concentration on the plant from overhead irrigation involves two separate stages. The first stage is determining the amount retained on plants after

being sprayed by irrigation water. The second stage is the calculation of the additional contamination as a result of root uptake and resuspension of contaminated soil onto the plant. The two stages are then added to obtain a combined contaminant concentration on edible plant surfaces. The plant concentration is then consumed according to each plant type, and a dose conversion factor is applied to the total intake to calculate the final dose from ingestion of produce.

In order to calculate the concentration on the plant following the initial deposition, an estimate must first be made of the deposition rate [Kennedy and Strenge, 1992]:

Equation 19

$$R = \{IR * r_v * T_v * C_w \} / Y_v$$

Where:

- R = Average deposition rate to edible parts of plant from application of irrigation water (pCi/kg*d)
- IR = Application rate of irrigation water (L/m²*d)
- r_v = Fraction of initial deposition retained on plant (dimensionless)
- T_v = Translocation factor for transfer of radionuclides from plant surfaces to edible parts (dimensionless)
- C_w = Average concentration in irrigation water (assumed constant) (pCi/L)
- Y_v = Plant yield (kg wet weight/m²)

Following the estimate of the deposition rate, a calculation of the contribution from direction deposition is an ordinary, first order, linear differential equation. The solution to the equation is as follows:

Equation 20

$$C_{plant} = R / \lambda \{1 - e^{-\lambda t} \}$$

Where:

- C_{plant} = The radionuclide concentration in the plant from deposition onto plant surfaces (pCi/kg)
- λ = Effective weathering and decay constant (d-1)
- t = growth period for plant (d)

For simplicity, losses from radiological decay during the holdup period[70] and consumption period are neglected. This conservative assumption has no significant impact on the dose contribution, as the radionuclides of interest have long half-lives.

[70] The holdup period is the time between produce harvest and consumption.

The second stage of the calculation is the estimate of the concentration in plants resulting from resuspension and root uptake. In order to estimate this contribution, the average soil concentration must first be calculated. This linear differential equation is similar to equation 20, with the exception of the loss term.

The loss of contaminants from soil is due to leaching by infiltrating water. This infiltration rate is different from the estimated infiltration rate of the buried waste of the LLRW disposal site, as the area of interest for plants (in our calculations) is the first 15 centimeters of soil (and not the five meters of soil needed to get to the buried waste). As a result of this decrease in the depth of interest (compared to the contaminated zone), infiltration rates may be significantly higher than the buried waste contaminated zone, yet not impact deeper depths, due to the large percentage of evaporation losses that are estimated to occur in the top 0.5 m of soil.[71]

Equations 21 through 24 are necessary in order to determine the loss of contaminants due to leaching [Yu, et al, 1993]. Equation 21 utilizes a combination of site-specific and default data to obtain an estimated infiltration rate.

Equation 21

$$I = \{1 - C_e\}\{\{1 - C_r\}P_r + I_{rr}\}$$

Where:

- I = Infiltration rate (m/year)
- C_e = Evapotranspiration coefficient (dimensionless)
- C_r = Runoff coefficient (dimensionless)
- P_r = Precipitation rate (m/year)
- I_{rr} = Irrigation rate (m/year)

In order to determine the retardation factor, it is first necessary to calculate the saturation ratio in equation 22.

Equation 22

$$R_s = \{I / K_{sat}\}^{1/\{2b+3\}}$$

Where:

- R_s = Saturation Ratio
- K_{sat} = Hydraulic conductivity (m/year)
- b = soil specific exponential parameter [Yu, et al, 1993][72] (dimensionless)

[71] Although the modeling assumed that the majority of plant root depth is 15 cm, it was observed that root depth was independent of the final equilibrium soil concentration, as the leach rate would be adjusted to the root volume, regardless of depth.

[72] The soil-specific b parameter is an empirical parameter used to evaluate the saturation ratio of the soil.

The retardation factor in equation 23 [Yu, et al, 1993] is the ratio of the pore water velocity to the radionuclide transport velocity.

Equation 23

$$R_d = 1 + \{\rho_b * K_d\} / \{p_t * R_s\}$$

Where:

- R_d = Retardation factor (dimensionless)
- ρ_b = Soil density (g/cm^3)
- p_t = Soil porosity (dimensionless)
- K_d = Distribution coefficient (cm^3/g)

Equation 24 [Yu, 1993] is used to obtain a time independent estimate of the leach rate in the top 15 centimeters of soil as a result of the application of irrigation water and local precipitation.

Equation 24

$$L = I / \{\theta * T * R_d\}$$

Where:

- L = Leach rate (y^{-1})
- θ = Volumetric water content (dimensionless)
- T = Thickness of contaminated zone (m)

Having obtained the information necessary to calculate the loss term in the soil, equation 25 [Kennedy and Strenge, 1992] calculates the radionuclide deposition rate onto the soil.

Equation 25

$$R_{soil} = \{C_w * IR\} / P_s$$

Where:

- R_{soil} = Average deposition rate onto soil (pCi/kg*d)
- P_s = Aerial soil density (kg/m^2)

The final concentration at the end of the growing period is shown in equation 26. In order to account for continued deposition over time, equation 26 was modified by taking the time for plant growth to infinity. The resulting equilibrium concentration is simply the deposition rate divided by the leach rate.

Equation 26

$$C_{soil} = R_{soil} / (L * 365) * \{1 - e^{-Lt}\}$$

Where:

- C_{soil} = Radionuclide soil concentration at end of growing period (pCi/kg)

Finally, equation 27 calculates the concentration in the plant due to uptake and resuspension [Kennedy and Strenge, 1992].

Equation 27

$$C_{plant} = \{ML + B\} * W_{d-w} * C_{soil}$$

Where:

- C_{plant} = Radionuclide concentration in plant (pCi/kg)
- ML = Mass loading factor for resuspension of soil to edible portions of plant (dry weight)
- B = Concentration factor for uptake of soil to plant (dry weight basis)
- W_{d-w} = Conversion factor for plants from dry weight to wet weight

The total contaminant concentration is the sum of equations 20 and 27. The formula is as follows:

Equation 28

$$Dose_{plants} = \frac{C_{plants}}{27} * Q_{plants} * DCF * F * 10^{8}$$

Where:

- $Dose_{plants}$ = Committed effective dose from ingesting contaminated vegetation (mrem/year)
- C_{plants} = Contaminant concentration in plants (pCi/g)
- Q_{plants} = Intake rate of vegetation (kg/year)
- DCF = 50 year committed effective dose conversion factor for ingestion of contaminants (Sv/Bq)
- F = Fraction of contaminated material that is grown
- 10,000,000 = Converts Sieverts (Sv) to mrem and grams to kilograms
- 27 = Converts pCi to Bq

The fraction of contaminated material that is assumed grown in a particular location is obtained from the EPA [U.S. EPA 1991]. To summarize, in a rural setting for the general population, the EPA assumes that 40% of all vegetables and 30% of all fruits

are grown by the individuals.[73] The basis for the EPA-recommended fractions is that while farm families can grow a large number of fruits and vegetables, it is unlikely that the individual (or family) could grow a sufficient variety to meet dietary needs and tastes.[74] For the Native American, it is assumed that 62% of the fruit and vegetables are grown locally [Harris and Harper, 1997].

4.5.1.2 Ingestion of Fruit and Vegetable Products Contaminated by Direct Removal of Contaminated Waste

The calculation of the onsite concentration in fruits and vegetables from direct contact with contaminated waste parallels the discussion of the analysis performed for the irrigation pathway, with a few exceptions. First, the soil concentration for the contaminated soil uncovered (from the drilling of a well) is the result of a single deposition event, as opposed to deposition over time in the irrigation pathway analysis. The contaminant concentration for the well material analysis is a maximum when initially deposited, and is reduced over time, due to leaching into the soil and radioactive decay. By comparison, the contaminant concentration for a particular contaminant in the irrigation pathway reaches an equilibrium value over a period of time, due to continued deposition, year after year. This equilibrium contaminant concentration for the irrigation pathway would remain so until irrigation activities cease. Only then would the irrigation pathway contaminant concentration resemble the reduction in contaminant concentration for the well volume material. Second, the plants in the irrigation pathway receive a portion of their contamination from direct deposition of the irrigation water (overhead spray is assumed). For the well volume material, the only pathway is root uptake and resuspension to the plants, as opposed to direct deposition as well (for irrigated plants).[75]

4.5.2 Ingestion of Meat and Dairy Products

The following pathways are considered in the analysis of animal ingestion:

- Ingestion of beef cattle
- Ingestion of milk (dairy cattle)
- Ingestion of poultry
- Ingestion of eggs

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[73] Due to the limited size of area assumed, grains are not assumed to be locally grown. There is also little evidence of individuals growing grain for personal and not commercial use.

[74] The EPA-recommended fraction is not based upon the size of land. For comparison, the NRC [U.S. NRC, 1977] assumes that an individual's entire diet is raised on a 10,000 m^2 site. NUREG/CR 3620 [Napier. et al, 1984] further defined the fractional breakout, roughly estimating that approximately 75% of the family's needs could be produced with land the size of the 2,500 m^2 plot. Based upon this information and the inability of a family to produce a sufficient variety of fruits and vegetables, the EPA values appear appropriate and sufficiently conservative.

[75] The 1,500 m^2 contaminated soil area for the well volume analysis is a portion of the same area that is used for the irrigation pathway. Although the analysis is performed separately, the results are summed, as the 1,500-m^2 area is expected to also contain contamination as a result of contaminated irrigation water.

The animals, in turn, are exposed to contamination via a number of mechanisms. The mechanisms considered are:

- Direct Ingestion of Well Water by Animals
- Animal Ingestion of Plants Contaminated Directly from Irrigation Spray and from Root Uptake and Resuspension of Soil Contamination[76]
- Direct Ingestion of Contaminated Soil

4.5.2.1 Direct Ingestion of Well Water by Animals

The computer code GWSCREEN [Rood, 1994] estimates the contaminant concentration in the groundwater. The groundwater concentration output is then directly used as the concentration in the well water that the animals drink. A transfer factor is then utilized to estimate the contaminant concentration in the edible portion of the animal as a result of ingesting contaminated well water. The formula for estimating the concentration in the animal product is as follows:

Equation 29

$$C_{animals,water} = C_W * Q_w * TF$$

Where:

- $C_{animals, water}$ = Concentration in animals due to water intake (pCi/kg)
- C_W = Groundwater concentration (pCi/l)
- Q_W = Intake rate of water by animals (l/d)
- TF = Transfer factor that takes into account the concentration in the edible portion of the animal to the concentration in the water (pCi/kg/pCi/d)

The contaminant intake amounts are located in the supporting documentation for this analysis [Thatcher, et al, 1998].

4.5.2.2 Ingestion of Plants Contaminated Directly from Irrigation Spray and from Root Uptake and Resuspension of Soil Contamination

The plants irrigated for the animals include fresh forage, stored hay, and stored grain. The specific intake of each fraction for an animal generally depends upon the season. However, an average ingestion amount for each animal per food group is utilized for these calculations [Kennedy and Strenge, 1992]. Specific values for each parameter are located in the supporting documentation for this analysis [Thatcher, 1998]. The methodology for the animal ingestion pathway closely follows that of direct plant ingestion (by humans). The main difference is that humans consume plant material at the end of the growing season, whereas animals consume the plants continuously.

[76] Animal contamination as a result of direct contamination of waste is not considered, due to the limited size of the material removed.

The calculation of the concentration on the plant involves two separate stages. The first stage is the calculation of the contamination on the plant as a result of directly deposited material. The second stage is the calculation of the additional contamination as a result of root uptake and resuspension. The two stages are then added to obtain a combined contaminant concentration on edible plant surfaces.

The first stage in the calculation of the concentration of the plant is an estimate of the deposition rate. The formula for the deposition rate [Kennedy and Strenge, 1992] is:

Equation 30

$$R = \frac{I_{rr} * r_v * T_v * C_w}{Y_v}$$

Where:

- R = Average deposition rate to edible parts of plant from application of irrigation water (pCi/kg*d)
- I_{rr} = Application rate of irrigation water (L/m²*d)
- r_v = Fraction of initial deposition retained on plant (dimensionless)
- T_v = Translocation factor for transfer of radionuclides from plant surfaces to edible parts (dimensionless)
- C_w = Average concentration in irrigation water (assumed constant) (pCi/l)
- Y_v = Plant yield (kg wet weight/m²)

Following the estimate of the deposition rate, a calculation of the contribution from direction deposition is a first-order linear differential equation. Equation 31 applies to stored grain and hay, as the formula takes into account the accumulation of contamination over the entire growing season. The solution to the equation is as follows:

Equation 31

$$C_{plant,stored} = R / \lambda \left\{1 - e^{-\lambda t}\right\}$$

Where:

- $C_{plant, stored}$ = The radionuclide concentration in the plant from deposition onto plant surfaces (pCi/kg)
- λ = Effective weathering and decay constant (d-1)
- t = growth period for plant (d)

For simplicity, losses during the holdup period[77] and consumption period are neglected. This conservative assumption has no significant impact on the dose contribution, as the three radionuclides of interest have long half-lives.

The calculation of the contribution from direct deposition for grasses (fresh forage) takes into account the fact that animals ingest the contaminated grass during the entire growing period. As a result, the amount of contamination ingested is an average of the entire growing period.[78] The solution for this equation is as follows:

Equation 32

$$C_{plant,direct,avg} = \frac{(\frac{R * t}{\lambda}) - (\frac{R}{\lambda^2} * (1 - E^{(-\lambda * t)}))}{t}$$

Where:

- $C_{plant,direct,avg.}$ = Average plant concentration for fresh forage (pCi/kg)

The second stage of the calculation is the estimate of the concentration in plants resulting from resuspension and root uptake. In order to estimate this contribution, the average soil concentration must first be calculated. This linear differential equation is similar to equation 31, with the exception of the loss term.

Prior to calculating the average soil concentration, the loss due to leaching must be estimated. The loss of contaminants from soil is due to leaching by infiltrating water. This infiltration rate is different from the estimated infiltration rate of the buried waste of the LLRW disposal site, as the area of interest for plants is the first 15 centimeters of soil. As a result of this decrease in the depth of interest (compared to the contaminated zone), infiltration rates may be significantly different than those of the deeper wastes due to increased evaporation losses and differences in soil density.

Equations 21 through 24 are used to determine the loss of contaminants due to leaching [Yu, et al, 1993]. Equation 33 [Kennedy and Strenge, 1992] calculates the radionuclide deposition rate onto the soil.

Equation 33

$$R_{soil} = \frac{C_w * I_{rr}}{P_s}$$

Where:

- R_{soil} = Average deposition rate onto soil (pCi/kg*d)
- P_s = Aerial soil density (kg/m²)

[77] The holdup period is the time between produce harvest and consumption.
[78] Equation 15 is derived by integrating equation 14 with respect to time, to yield an average value.

The final concentration at the end of the growing period is shown in equation 34. In order to account for continued deposition over time, equation 34 was modified by taking the time for plant growth to infinity. The resulting equilibrium concentration is simply the deposition rate divided by the leach rate.

Equation 34

$$C_{soil} = \frac{R_{soil}}{L * \left\{1 - e^{-L}\right\}}$$

Where:

- C_{soil} = Radionuclide soil concentration at end of growing period (pCi/kg)

Finally, equation 35 calculates the concentration in the plant due to uptake and resuspension [Kennedy and Strenge, 1992]:

Equation 35

$$C_{plant,uptake+resuspension} = \left\{ML + B\right\} * W_{d-w} * C_{soil}$$

Where:

- C_{plant} = Radionuclide plant concentration (pCi/kg)
- ML = Mass loading factor for resuspension of soil to edible portions of plant
- B = Concentration factor for uptake of soil to plant (dry weight basis)
- W_{d-w} = Conversion factor for plants from dry weight to wet weight

Once the estimated animal feed concentrations have been calculated (equations 31, 32, and 35), the concentration in the edible portion of the animal may then be estimated. The formula for estimating the contribution in the animal due to deposition and uptake from fresh forage is:

Equation 36

$$C_{Animals,forage} = (TF * Q_{a,forage} * f_w) * (C_{plant,direct} + C_{plant,uptake+resuspension})$$

Where:

- $C_{Animals, forage}$ = Concentration in animals as a result of ingesting contaminated fresh forage
- TF = Transfer factor relating the concentration in the edible portion of the animal to the intake concentration (pCi/kg/pCi/d)
- $Q_{a, forage}$ = Consumption rate of fresh forage by animals (Kg/d)
- f_w = Fraction of forage that is contaminated (unitless, 1)

The formula for estimating the concentration in the edible portion of the animal as a result of ingesting stored feed is as follows:

Equation 37

$$C_{Animal,storedfeed} = TF * ((f_w * C_{grain} * Q_{a,grain}) + (f_w * C_{storedhay} * Q_{a,storedhay}))$$

Where:

- $C_{animal,\ stored\ feed}$ = Concentration in animals as a result of ingesting stored feed (pCi/kg)
- C_{grain} = Concentration in the grain (pCi/kg)
- $C_{stored\ hay}$ = Concentration in the stored hay (pCi/kg)
- $Q_{a,\ grain}$ = Consumption rate of grain by the animal (kg/d)
- $Q_{a,\ stored\ hay}$ = Consumption rate of stored hay by the animal (kg/d)

4.5.2.3 Ingestion of Soil by Animals

Animals inadvertently ingest soil in the process of consuming feed. For this process, the animals are presumed to only ingest soil while consuming fresh forage. The amount of soil ingested is taken to be a fraction of the amount of forage consumed. The formula for the concentration in the edible portion of the animal as a result of ingesting contaminated soil is [Kennedy and Strenge, 1992]:

Equation 38

$$C_{Animals,soil} = TF * f_w * Q_{a,forage} * IF * W_{D-W} * C_{Soil,avg}$$

Where:

- $C_{animals,\ soil}$ = Concentration in animals due to the ingestion of soil (pCi/kg)
- $Q_{a,\ forage}$ = Consumption rate of vegetation by animals (kg/d)
- IF = Intake fraction of soil (unitless)
- W_{D-W} = Dry to wet weight conversion factor (unitless)
- $C_{soil,\ ave}$ = Average contaminant concentration in soil (pCi/kg)

4.5.2.4 Overall Contribution from the Animal Pathway

Equations 29, 36, 37, and 38 are combined to obtain an overall contribution for the animal pathway from the ingestion of groundwater well, plants, and soil. The resulting estimated dose is:[79]

Equation 39

[79] Note that the equation is simplified by assuming that no decay occurs during the period of time between harvest and consumption. This assumption is valid, as the radionuclides of interest for the groundwater pathway are very long lived.

$$D_{Animalpathway}^{Humans} = DCF * 365d/y * Q_{h,animalproduct} * \frac{10^5}{27} * (C_{Water}^{Animals} + C_{stored}^{Animals} + C_{forage}^{Animals} + C_{Soil}^{Animals})$$

Where:

- D^{humans} = Dose to humans from the animal ingestion pathway (mrem/year)
- DCF = Dose conversion factor (Sv/Bq)
- $\frac{10^5}{27}$ = Factors to convert Sv to mrem and pCi to Bq
- $Q_{h,\ animal\ product}$ = Consumption rate of specific animal product by humans (kg/d)

4.6 Surface Water

Surface water on or in the near vicinity of the LLRW disposal site does not exist. Scenarios involving surface water are therefore not used for this analysis.

5.0 Estimated Offsite Dose

The Proposed Action and each alternative have been analyzed for the Rural Resident and Native American scenarios to determine offsite risk. Methods discussed in Section 4 were used for the analysis. The results of the analyses are presented in terms of the maximum expected dose and incremental lifetime cancer risk. The following bullets are a brief summary of the conditions that apply to the analyses; further details can be located in Sections 3 and 4:

- Groundwater-related contributions include drinking water ingestion, food ingestion, and other related pathways such as sweat lodge inhalation for Native Americans.

- All groundwater results represent the maximum downgradient location (i.e., the maximum concentration for onsite or offsite).

- Radionuclides modeled for groundwater dose are H-3, C-14, Tc-99, Cl-36, I-129, U-234, U-235, U-238, Pu-238, and Pu-239 (see the Groundwater Analysis report in the FEIS for further discussion on the derivation of the contaminant concentration).

- All results other than groundwater relate to the diffusion or dispersion of contaminated soils or gases from onsite sources.

- All calculations assumed the loss of institutional controls at 107 years.[80]

[80] 107 years represent 100 years of institutional controls and seven years of onsite "active" maintenance.

- The results tables contain a segregation at 500 years. This time break is a result of the increased contribution from sealed radium sources that are assumed to contribute to dose after 500 years.

- Tritium with a 12.3-year half-life will decay significantly prior to the end of the institutional control period. All impacts from tritium are less than 250 years following closure.

- Results are only calculated for radionuclides with travel times less than 10,000 years.

- Total dose is calculated by the sum of groundwater-related activities and diffusion of gases and dust from onsite. Dose is then multiplied by the assumed years of exposure and a probability of fatal cancer coefficient [ICRP, 1990]. The probability coefficient is .0005/rem effective dose equivalent. The Rural Resident Adult risk calculations are based upon 30 years of exposure. The Native American Adult risk calculations are based upon 70 years of exposure. The Rural Resident Child risk calculations are based on 6 years of exposure as a child and 24 years of exposure as an adult. The Native American Child risk calculations are based on 6 years of exposure as a child and 64 years of exposure as an adult.

- Dose conversion factors from ICRP 72 [ICRP, 1995] are used for this report, as it is the only reference that segregates the dose conversion factors based upon age, thereby allowing for a more accurate assessment of the potential exposure to a child.

- Spreadsheet results containing detailed calculations are located in supporting documentation [Thatcher, et al, 1998].

5.0.1 Differences from the DEIS Analysis

- The FEIS differs from the Draft Environmental Impact Statement (DEIS) for this LLRW facility in a few significant ways, namely: Radium analysis. The radium analysis was improved in a number of methods in an attempt to more accurately quantify the potential dose contribution.
  1. The radium waste was segregated by depth based upon analysis that M. Elsen provided [Elsen, 2003]. In the DEIS, the radium activity for each closure time period was assumed to be homogenized throughout the entire waste volume and then analyzed from the middle of the active trench volume. For the FEIS, the radium waste was segregated into 3 feet, 8 feet, 16.5 feet, and 23 feet in depth based upon when the waste was disposed, and the disposal practices at the time of disposal. This waste segregation has a tremendous impact on the predicted dose, as a significant fraction of recent and future waste is disposed near the bottom of the trenches, as opposed to closer to the surface.

2. Segregation of future waste. Based upon the practices outlined in the Elsen memo, future waste is segregated assuming a 90% at greater than 23', and 10% at greater than 8' split.
3. The DEIS projected that 1.69 Ci/yr of radium will be disposed onsite. The FEIS assumes that 4.294 Ci/yr of radium waste is disposed on the site. As the analysis in the FEIS shows, the impact of the increased waste is significantly diminished due to the waste segregation discussed above.
4. The moisture saturation fraction for the site soils was modified to more accurately reflect the average soil characteristics for the site and surrounding area, as opposed to using the most conservative values.
5. The asphalt and Composite GCL covers were assumed to limit almost all radon emanation in the first 500 years following closure, due to the design of those cover materials and limited permeability. The Enhanced covers (bentonite, Composite GCL, asphalt) and the proposed covers were assumed to degrade in performance 500 years after closure, to account for increased porosity of the cover material due to subsidence and material degradation.

The original analysis for this FEIS segregated the site closure into three separate timeframes. In each of those timeframes, the average concentration for each contaminant was determined by taking the total curies of waste and dividing by the total volume of waste plus fill for each closure timeframe. The net effect of this action was to dilute the overall concentration for a given contaminant, as the initial waste and corresponding fill volume was highest for the 2005 closure period, and lower for subsequent closure periods. One limitation in this assumption is that while it is true that the overall average concentration of the waste is less for the 2056 closure (or 2215) as compared to the 2005 closure, it ignores the fact that the higher 2005 concentration does still exist on the site regardless of the closure date.[81] As a result of the artificially diluted contaminant concentrations, the assumption was made that the intruder would locate in the same original waste location and therefore be exposed to the same waste concentration (accounting for decay over time for the various closure dates). The impact of this assumption is particularly evident in the Composite GCL covers for the three closure dates and is discussed more fully in the following section.

5.0.2 Sweat Lodge Impacts

The potential impact as a result of the use of the sweat lodge merits specific attention. The operating assumption is that 100% of the contaminants in the groundwater (used as the source of steam for the sweat lodge) will become airborne and remain available for inhalation. Uranium and plutonium compounds have a higher melting point than the temperature observed in a sweat lodge and must be entrained in the water transitioning to steam to be available for inhalation. Of those contaminant particles in the air, it is likely that the deposition rate will be higher than that for water vapor and would also serve to decrease the average air concentration. In addition, it is likely that a fraction of the contaminants will fail to become entrained in the water and become airborne,

[81] The higher concentration would directly impact the radon flux estimates and the available activity unearthed by the well driller.

further reducing the air concentrations from those used in the calculation. The sweat lodge calculations are therefore considered a worst case estimate of the potential exposure to contaminants. Until data are available on the potential air concentration in a similar environment, the current model is considered the appropriate method for estimating exposure.

For the all of the covers with the exception of the site soils and late installed 2056 cover, the sweat lodge contribution via the inhalation pathway accounts for about 85% of the groundwater related dose to the Native American adult in the 500 to 1,000-year timeframe, and over 60% of the dose in the greater than 1,000-year time frame. Sweat lodge inhalation doses account for over 90% of the peak contributions for the less than 500-year exposure for the site soils cover. In the enhanced cover installed late, sweat lodge related exposures account for approximately 70% of the peak dose for the less than 500-year timeframe. Perhaps as a summary, little differences would exist for the Rural Resident and Native American exposure scenarios were it not for the large dosimetric contribution as a result of contaminants used in a sweat lodge.

5.0.3 Separate Radium and Cesium Impact Analysis

In the summary tables for dose, it was previously mentioned that the assumption was made that the intruder would locate (drill a well and build a home) in the area of the initial waste deposition, as the average radium concentration for the assumed homogenized waste was greater than for the waste from other locations at later disposal time periods. The analysis in this section will show that this assumption is conservative.

For all future waste, 4.294 Ci/yr of radium is assumed to be disposed at the site. Considering the 51-year difference between the 2005 and 2056 closure date, this amounts to a decay corrected value of 216.6 Ci of waste. Ninety percent of this waste is presumed to be disposed at the bottom of the trench (>23'), and 10% is presumed to be disposed at greater than 8 feet in depth. This additional waste is divided into the additional waste plus volume for the 2056 closure period.[82] In this comparative analysis, the closure time period selected is the 2056 closure, and the cover is the Composite GCL. The analysis displayed in Table 5.1.1 estimates the dose contribution from radon at 84 mrem/y for the Rural Resident Adult. In comparison, by modifying the parameters to show the impact of only the location of the site that contains the post-2005 waste until closure, the relative radon impact is 17 mrem/y. The 90% of the waste buried at greater than 23 feet contributes about 2 mrem/y to the onsite intruder. The remaining 15 mrem/y contribution is from the waste buried at 8 feet or greater. In order for the future waste to be comparable in dose to the analysis presented in Table 5.1.1, 75% of the future waste would have to be buried at this significantly shallower depth.

In the DEIS for the LLRW facility, the analysis assumed complete homogeneity of the waste for each of the LLRW facility's closure time periods assumed. Applying this

[82] The total closure volume plus fill for 2056 is 1.08 E+12 grams; for 2005 the value is 7.58 E+11 grams; the difference of 3.22 E+11 is the additional waste plus fill volume.

methodology to the additional waste used for the future waste as compared to the original DEIS activity results in a predicted activity of 63 mrem/y as compared to the current analysis in Table 5.1.3 of 84 mrem/y.

In summary for the radium analysis and potential impact, the radium and resulting radon (plus progeny) contribution to dose is analyzed in three different methods and shows that the current assumption of the intruder only accessing the original waste is conservative. The radium analysis also shows that it would take over 75% of all future waste (at 4.3 Ci/yr) buried at 8 feet (as opposed to 90% at greater than 23 feet) to result in a dose contribution that equals the current analysis contribution of the intruder accessing the original waste volumes. The groundwater analysis concludes that radium is not a contributor via the water pathway. Diffusion of radon from onsite to offsite environs is less than 1 mrem/y, even for locations close to the LLRW site boundary. Significant onsite radon contributions are limited to the intruder's building a home with a basement, as homes built without basements would have dose contributions less than a tenth of the current analysis.

Recalling that the analysis assumes that the intruder accesses the same waste for all three time periods, the only differences in concentration are therefore due to decay of the waste. Cesium 137 is the only radionuclide with a short half-life and is the majority contributor of dose onsite in the near term (<500 years)[83]. With a 30-year half-life, even the 51-year difference in closure dates for the 2005 and 2056 closure periods results in a significant decay (~25 to 30 mrem/y depending on the scenario) of the cesium source term. Reviewing the 2215 intruder dose contribution and comparing these to the 2056 closure date reveals an additional 14 mrem/y of decay of the cesium source term. These near term differences in analytical results matter little, as the less than 500-year time period is not the maximum dose period for analysis. Although it is possible that future Cs-137 activities and resulting concentrations may equal or slightly exceed the predicted concentrations from the earlier waste disposal areas, the radium contributions to dose far exceed the Cs-137 contributions and are the primary driver in the decision to locate the intruder in the same location onsite.

5.1 Onsite and Offsite Results

This section presents the results for all of the proposed alternatives for the six scenarios identified in Section 3. Table 5.1.1 is the summary of estimated dose and is segregated into three different time increments. In order to simplify discussion, the discussion for both onsite and offsite results will be reviewed by the different alternatives. Table 5.1.2 is the estimated dose converted to risk. Table 5.1.3 presents the groundwater related contribution to dose.

5.1.1 Proposed US Ecology Cover 2056

[83] In addition, a comparative analysis was performed for the pre 2005 and post 2005 waste activity. The pre-2005 concentration of waste exceeded the post 2005 concentration for a majority of radionuclides.

When considering the various alternatives, the US Ecology proposed cover provides the lowest predicted dose for the offsite scenarios and an onsite dose less than the 100 mrem/y limit. Groundwater-related contributions provide over 95% of the dose to the offsite scenarios, with tritium contributing 40% of the 18 mrem/y to the Native American Adult for the time period less than 500 years. All other scenarios receive less than half of the predicted exposure that the Native American Adult is estimated to receive. This increased exposure is due to the sweat lodge contributions that are more fully discussed at the end of this section. The Proposed US Ecology cover, while not as robust in design as some of the enhanced covers, allows for a greater amount of contaminants to leach out of the waste prior to cover failure. Therefore, when the cover does fail, the peak concentrations for contaminants are not as great, as a significant amount of leaching has already occurred (as compared to the enhanced covers). So, while the Proposed US Ecology cover provides a lower predicted peak dose (predominately from groundwater), a greater amount of leachate contaminant is in the groundwater over a longer period of time.

Over 60% of the onsite intruder dose is caused by radon (and progeny) in the home. The remaining contributions are caused by the resultant exposures from the well drilling, unearthing waste and bringing it to the surface, and groundwater-related contamination.

5.1.2 Enhanced Asphalt, Bentonite, and GeoSynthetic/GCL Cover 2056

All three enhanced covers are reviewed together, as their performance characteristics are very similar. From a groundwater mobility perspective, the three covers are considered to behave the same in terms of cover failure timeframe and water infiltration during the period the cover is considered "intact". For a more in-depth discussion of the groundwater analysis, please refer to the Groundwater Appendix of the FEIS.
For the offsite analysis, the Native American Adult receives an estimated 22 mrem/y peak dose for the greater than 1,000 year timeframe, with over 60% of the contribution stemming from sweat lodge inhalation exposure. All other scenario exposures are less than 10 mrem/y.

The onsite intruder analysis reveals again that radon (and progeny) contribute over 60% of the dose. All three covers are considered to perform very similarly in terms of radon emanation, with a few exceptions. In the 0 to 500 year timeframe, both the Enhanced Asphalt and the Enhanced GeoSynthetic/GCL Cover are considered to inhibit almost all radon emanation. As time progresses, the GeoSynthetic/GCL Cover contains a slightly lesser amount of clay as compared to the Enhanced Bentonite Cover. This clay is considered to be the only remaining barrier for the GeoSynthetic/GCL Cover following failure of the HDPE and, as a result, may provide a slightly greater radon dose to the intruder. In comparison to the Thick Homogeneous Cover, all three covers will retain some protective benefit as a result of the additional protective barrier and will result in a long-term continued performance for protection

against radon emanation, although in a degraded condition.[84] It should be noted that the predicted results for all three enhanced covers are sufficiently close that no single cover, from a predictive dose standpoint, could be singled out as clearly outperforming the other enhanced covers. While the Asphalt and Bentonite covers' estimated onsite doses are less than the 100 mrem/y limit, it would be difficult to base cover acceptability upon these results alone, due to the large uncertainty associated with the radon emanation estimates in a home intruder setting.

The Native American Upland Hunter Scenario is probably the most realistic intruder scenario when one considers the fact that this LLRW site is located within the 200 Area of the Hanford Site. This location effectively prevents any long-term intruder habitation from occurring, leaving limited onsite scenarios such as the Upland Hunter as the only viable intruder scenario. The Upland Hunter receives a dose contribution from drinking water ingestion due to the contaminated water that is carried with him/her; the remaining dose is a result of outdoor radon exposure. A predicted 1-mrem/y dose to the Native American adult Upland Hunter is for a seven-day hunting trip. The Native American child is estimated to receive 2 mrem/y, slightly greater than that of the adult, which can be attributed to increased uptake rates for contaminants from drinking water intake.

The Native American Columbia River Subsistence Resident scenario is included in the analysis, and the predicted dose almost matches the results of the Native American immediately offsite of the LLRW facility. Section 3 of this appendix further discusses the details of the scenario. Multiple layers of conservatism are included in the assumption that the seeps along the river would contain concentrations similar to the predicted concentrations immediately beneath the LLRW facility. A single correction to the predicted seep concentration involves accounting for riverbank dilution in the observed seeps water [Guensch, G.R & Richmond, M.C., 2001]. In addition, it is not realistic to assume that a subsistence resident can sustain all of the supporting pathways with the volumes currently observed from seeps. Limited confidence should be placed on the estimated 11-mrem/y to the Native American adult for this scenario, other than to say that any Columbia River scenario would certainly result in exposures well less than the 25 mrem/y limit.

5.1.3 Enhanced GeoSynthetic/GCL Cover 2005 and 2215

The GeoSynthetic/GCL Cover is analyzed for both immediate closure as well as a filled site closure in the year 2215. The cover is the same as is analyzed in 2056, with the only difference being the source term. However, because the initial source term in the first 40 years of site activity indicated a higher activity (particularly for radium) than the calculated concentrations for future year disposals, the conservative assumption was made that the intruder would access only the higher activity portion of the site. The estimated impact from the three closure dates varies little as a result.

[84] The impact of a degraded barrier for radon was modeled by increasing the gaps and voids of the clay or asphalt layer, such as might occur over time due to settlement.

Table 5.1.3 also shows that the estimated impact from groundwater- related contributions is essentially the same for all three closure periods (2005, 2056, and 2215). The open trench for the first 40 years makes a large difference when the endpoint is the maximum concentration/dose for different cover scenarios because essentially, all covers perform the same for the first 40 years, when releases are the highest. The post-1,000-year groundwater contribution of 21 mrem/y for the 2005 closure period is slightly less than the 24 mrem/y for the 2215 closure (or 22 mrem/y for the 2056 closure). The slight differences can be due to the increases over time in the overall source term. This source term impact from the groundwater pathway is in contrast to the impact to the intruder from the radium or well volume material as a result of drilling a well. For the intruder, these actions are location-specific, whereas the groundwater impact does not depend upon the location within the site, but instead on the activity disposed at the site. Both covers are less than the offsite limit of 25 mrem/y.

The intruder analysis predicts a peak dose of 107 mrem/y for the Native American Adult for the 2005 closure, and 101 mrem/y for the 2215 closure for the 500 to 1,000 year timeframe. A closure inspection of Table 5.1.3 indicates that the doses remain almost the same for the Native American Adult, yet decrease for the Rural Resident Adult and Child when comparing the 500 to 1,000 and greater than 1,000-year time periods. The roughly 9 mrem/y decrease for the Rural Resident scenarios is due to a ~13 mrem/y decrease in the radon contributions, due to radium decay and a small increase in the predicted groundwater concentrations. The Native American Adult has a larger groundwater increase for the same contaminant concentration increase (due to sweat lodge contributions) and the corresponding radon decrease.

5.1.4 Site Soils Cover 2056

The Site Soils Cover is a simplistic alternative that lacks any special barriers for water infiltration and is missing the improved soils used in a vegetative cover.[85] As a result, the onsite exposure estimates are significantly greater than for any other cover.

Table 5.1.3 provides the groundwater results and shows that the immediate impact on the groundwater is observed in the 0 to 500-year timeframe. Table 5.1.3 shows that the estimated groundwater contribution to the Native American Adult is 80 mrem/y. Seventy percent of the estimated 70 mrem/y from plutonium and uranium is due to inhalation while in the sweat lodge. The Native American child is exposed to a significantly lower extent to the limited time spent in a sweat lodge. The Native American Child also receives an offsite exposure greater than the 25-mrem/y limit, at 29 mrem/y.

The offsite analysis (Table 5.1.1) shows that the majority of the estimated 384 mrem/y to the Rural Resident Adult intruder is due to radon contributions, as the cover material lacks any significant mechanism to reduce the emanation rate.

[85] A vegetative cover is included in all other cover designs.

5.1.5 Enhanced Late GeoSynthetic/GCL Cover 2056

This cover matches the other GCL covers, with the exception that no trenches are covered (other than backfill to grade) until closure in 2056. As a result, the buried waste is open to significantly greater infiltration prior to the installation of an enhanced cover. Tables 5.1.1 and 5.1.3 display the impact of the delay. Predicted groundwater contaminant contributions of 130 mrem/y to the Native American Adult are significantly greater than the regulatory limit and greater than all other alternative covers analyzed. All other onsite scenarios exceed the 25-mrem/y regulatory limit as well. The results in the table also indicate that, following the initial 500 years after closure, contaminant concentrations reduce to levels less than the regulatory limit, but onsite contributions from radon (and progeny) would increase, partially offsetting the overall reduction in dose over time for the onsite intruder.

5.1.6 Homogeneous Cover 2056

The homogeneous cover is essentially a Site Soils cover with a five-foot vegetative cover placed on top.

From a groundwater perspective, this cover is assumed to perform as well as the enhanced covers in terms of limiting water infiltration. The offsite dose for all scenarios is slightly less than the 25 mrem/y offsite limit and matches the predicted offsite doses for the enhanced covers.

The onsite intruder results are exceeded for all scenarios primarily due to the increased radon emanation as compared to the other enhanced covers. Unlike the enhanced covers, no additional barrier is provided to limit gas emanation. In comparison to the Site Soils Cover, the Homogeneous Cover is thicker and does result in a reduced emanation rate and a correspondingly lower radon dose.

Table 5.1.1

Dose Estimate for All Covers and Scenarios (mrem/y)

d GeoSynt 2005	1000y-10,000y		93	91	104	89		6	7	21	11						
Enhanced GeoSynthetic 2215	0-500		28	25	44	30		8	8	18	9						
	500-1000		97	94	101	88		2	2	11	3						
	1000y-10,000y		88	86	101	86		7	9	24	14						

	Cover	Enhanced GeoSynthetic with Construction in 2056				
	Closure Date	2056				
	Timeframes (y)	0-500	500-1000	1000y-10,000y	0-500	500-1000
Scenarios	**Onsite Resident Intruder**					
	Rural Resident Adult	70	105	92	70	106
	Rural Resident Child	68	102	89	63	103
	Native American Adult	171	104	95	90	107
	Native American Child	82	95	87	71	95
	Offsite Resident					
	Rural Resident Adult	36	2	5	9	3
	Rural Resident Child	39	2	6	9	2
	Native American Adult	130	7	13	19	11
	Native American Child	48	3	9	10	4
	Onsite Upland Hunter					
	Native American Adult					
	Native American Child					
	Resident River					
	Native American Adult					
	Native American Child					

Dose Estimate for All Covers and Scenarios (mrem/y)

Cover	Enhanced Asphalt			Enhanced Bentonite			Enhanced GeoSynthetic		
Closure Date	2056			2056			2056		
Timeframes (y)	0-500	500-1000	1000y-10,000y	0-500	500-1000	1000y-10,000y	0-500	500-1000	1000y-10,000y
Scenarios									
Onsite Resident Intruder									
Rural Resident Adult	41	87	78	55	82	74	42	105	93
Rural Resident Child	36	83	76	50	79	72	37	102	91
Native American Adult	58	91	92	70	88	89	59	107	105
Native American Child	43	79	77	55	75	74	44	95	90
Offsite Resident									
Rural Resident Adult	8	2	6	8	2	6	8	2	6
Rural Resident Child	8	2	8	8	2	8	8	2	8
Native American Adult	18	11	22	18	11	22	18	11	22
Native American Child	9	3	12	9	3	12	9	3	12
Onsite Upland Hunter									
Native American Adult							0	1	1
Native American Child							0	1	2
Resident River									
Native American Adult							9	8	11
Native American Child							4	1	5

Table 5.1.1

Dose Estimate for All Covers and Scenarios (mrem/y)

Scenarios	Cover	Proposed US Ecology			Site Soils			Homogeneous		
	Closure Date	2056			2056			2056		
	Timeframes (y)	0-500	500-1000	1000y-10,000y	0-500	500-1000	1000y-10,000y	0-500	500-1000	1000y-10,000y
	Onsite Resident Intruder									
	Rural Resident Adult	56	87	78	132	384	214	76	173	147
	Rural Resident Child	51	84	75	130	382	212	71	169	145
	Native American Adult	72	94	87	190	336	195	88	164	151
	Native American Child	56	81	77	132	333	190	73	152	136
	Offsite Resident									
	Rural Resident Adult	8	3	6	17	5	7	8	3	7
	Rural Resident Child	8	3	7	20	7	8	8	4	9
	Native American Adult	18	12	16	81	7	11	18	12	23
	Native American Child	9	4	11	29	9	11	9	5	13
	Onsite Upland Hunter									
	Native American Adult									
	Native American Child									
	Resident River									
	Native American Adult									
	Native American Child									

Table 5.1.1

Table 5.1.2

Lifetime Cancer Risk

Scenarios	Cover	Enhanced GeoSynthetic with no cover until 2056			Enhanced GeoSynthetic			Enhanced GeoSynthetic		
	Closure Date	2056			2005			2215		
	Timeframes (y)	0-500	500-1000	>1000y	0-500	500-1000	>1000y	0-500	500-1000	>1000y
	Onsite Resident Intruder									
	Rural Resident Adult	1.04E-03	1.57E-03	1.37E-03	1.05E-03	1.59E-03	1.40E-03	4.15E-04	1.46E-03	1.32E-03
	Rural Resident Child	1.04E-03	1.56E-03	1.37E-03	1.03E-03	1.58E-03	1.39E-03	4.06E-04	1.45E-03	1.31E-03
	Native American Adult	2.56E-03	1.55E-03	1.43E-03	1.34E-03	1.61E-03	1.56E-03	6.64E-04	1.51E-03	1.52E-03
	Native American Child	2.30E-03	1.53E-03	1.41E-03	1.29E-03	1.57E-03	1.52E-03	6.23E-04	1.47E-03	1.47E-03
	Offsite Resident									
	Rural Resident Adult	5.37E-04	2.91E-05	7.41E-05	1.29E-04	4.82E-05	9.11E-05	1.16E-04	3.23E-05	1.11E-04
	Rural Resident Child	5.47E-04	3.01E-05	7.75E-05	1.31E-04	4.51E-05	9.39E-05	1.17E-04	3.30E-05	1.16E-04
	Native American Adult	4.54E-03	2.52E-04	4.43E-04	6.56E-04	3.83E-04	7.49E-04	6.24E-04	3.76E-04	8.42E-04
	Native American Child	4.29E-03	2.40E-04	4.33E-04	6.29E-04	3.61E-04	7.18E-04	5.98E-04	3.54E-04	8.11E-04
	Onsite Upland Hunter									
	Native American Adult									
	Native American Child									
	Resident River									
	Native American Adult									
	Native American Child									

Table 5.1.2

Lifetime Cancer Risk

Cover	Enhanced Asphalt			Enhanced Bentonite			Enhanced GeoSynthetic		
Closure Date	2056			2056			2056		
Timeframes (y)	0-500	500-1000	>1000y	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Scenarios: Onsite Resident Intruder									
Rural Resident Adult	6.10E-04	1.30E-03	1.17E-03	8.23E-04	1.23E-03	1.12E-03	6.24E-04	1.58E-03	1.40E-03
Rural Resident Child	5.96E-04	1.29E-03	1.16E-03	8.09E-04	1.22E-03	1.11E-03	6.10E-04	1.57E-03	1.39E-03
Native American Adult	8.72E-04	1.37E-03	1.38E-03	1.05E-03	1.32E-03	1.34E-03	8.86E-04	1.61E-03	1.57E-03
Native American Child	8.26E-04	1.33E-03	1.34E-03	1.01E-03	1.28E-03	1.29E-03	8.40E-04	1.57E-03	1.53E-03
Offsite Resident									
Rural Resident Adult	1.14E-04	3.27E-05	9.46E-05	1.16E-04	3.18E-05	9.39E-05	1.17E-04	3.36E-05	9.53E-05
Rural Resident Child	1.16E-04	3.34E-05	9.87E-05	1.18E-04	3.25E-05	9.80E-05	1.18E-04	3.43E-05	9.94E-05
Native American Adult	6.20E-04	3.77E-04	7.70E-04	6.25E-04	3.75E-04	7.70E-04	6.26E-04	3.79E-04	7.74E-04
Native American Child	5.93E-04	3.54E-04	7.40E-04	5.99E-04	3.53E-04	7.40E-04	6.00E-04	3.57E-04	7.43E-04
Onsite Upland Hunter									
Native American Adult							1.58E-05	4.12E-05	4.74E-05
Native American Child							1.59E-05	4.12E-05	4.85E-05
Resident River									
Native American Adult							3.12E-04	2.97E-04	3.71E-04
Native American Child							2.97E-04	2.74E-04	3.54E-04

Table 5.1.2

Lifetime Cancer Risk

	Cover	Proposed US Ecology			Site Soils			Thick Homogeneous		
	Closure Date	2056			2056			2056		
	Timeframes (y)	0-500	500-1000	>1000y	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Scenarios	**Onsite Resident Intruder**									
	Rural Resident Adult	8.40E-04	1.31E-03	1.16E-03	1.98E-03	5.76E-03	3.21E-03	1.14E-03	2.59E-03	2.21E-03
	Rural Resident Child	8.27E-04	1.30E-03	1.16E-03	1.98E-03	5.76E-03	3.20E-03	1.13E-03	2.58E-03	2.20E-03
	Native American Adult	1.08E-03	1.41E-03	1.31E-03	2.85E-03	5.04E-03	2.92E-03	1.32E-03	2.46E-03	2.26E-03
	Native American Child	1.03E-03	1.37E-03	1.28E-03	2.68E-03	5.03E-03	2.91E-03	1.27E-03	2.43E-03	2.22E-03
	Offsite Resident									
	Rural Resident Adult	1.18E-04	3.78E-05	8.63E-05	2.57E-04	7.73E-05	9.94E-05	1.21E-04	5.10E-05	1.10E-04
	Rural Resident Child	1.19E-04	3.93E-05	9.03E-05	2.66E-04	8.18E-05	1.03E-04	1.22E-04	5.17E-05	1.14E-04
	Native American Adult	6.41E-04	4.24E-04	5.69E-04	2.83E-03	2.42E-04	3.69E-04	6.36E-04	4.20E-04	8.07E-04
	Native American Child	6.13E-04	4.01E-04	5.53E-04	2.67E-03	2.50E-04	3.69E-04	6.09E-04	3.97E-04	7.76E-04
	Onsite Upland Hunter									
	Native American Adult									
	Native American Child									
	Resident River									
	Native American Adult									
	Native American Child									

Table 5.1.3 Groundwater-Related Dose by Scenario and Cover Type

	mrem/y	C-14 ONSITE			C-14 OFFSITE		
Scenario	Timeframes	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Rural Resident Adult	site soils 2056	0.1	0.1	1.5	0.1	0.0	1.5
	enhanced cover 2003	0.0	0.0	0.8	0.0	0.0	0.8
	enhanced cover 2056	0.0	0.0	1.0	0.0	0.0	1.0
	enhanced cover 2056 late	0.1	0.0	0.9	0.1	0.0	0.9
	enhanced cover 2215	0.1	0.1	1.7	0.0	0.0	1.6
	proposed cover 2056	0.0	0.1	1.1	0.0	0.0	1.0
Rural Resident Child	site soils 2056	0.1	0.1	1.7	0.1	0.0	1.6
	enhanced cover 2003	0.0	0.0	0.9	0.0	0.0	0.9
	enhanced cover 2056	0.0	0.1	1.1	0.0	0.0	1.1
	enhanced cover 2056 late	0.1	0.0	1.0	0.1	0.0	1.0
	enhanced cover 2215	0.1	0.1	1.8	0.0	0.0	1.8
	proposed cover 2056	0.0	0.1	1.2	0.0	0.0	1.1
Native American Adult	site soils 2056	0.3	0.1	2.6	0.2	0.0	2.5
	enhanced cover 2003	0.1	0.1	1.4	0.0	0.0	1.3
	enhanced cover 2056	0.1	0.1	1.7	0.0	0.0	1.6
	enhanced cover 2056 late	0.3	0.1	1.6	0.2	0.0	1.5
	enhanced cover 2215	0.2	0.2	2.9	0.0	0.0	2.7
	proposed cover 2056	0.1	0.1	1.8	0.0	0.0	1.7
Native American Child	site soils 2056	0.2	0.1	1.8	0.1	0.0	1.7
	enhanced cover 2003	0.1	0.1	0.9	0.0	0.0	0.9
	enhanced cover 2056	0.1	0.1	1.2	0.0	0.0	1.1
	enhanced cover 2056 late	0.2	0.1	1.1	0.1	0.0	1.0
	enhanced cover 2215	0.2	0.2	2.0	0.0	0.0	1.8
	proposed cover 2056	0.1	0.1	1.2	0.0	0.0	1.1

Table 5.1.3 Groundwater Related Dose by Scenario and Cover Type

	mrem/y	Cl-36			Tc-99		
Scenario	Timeframes	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Rural Resident Adult	site soils 2056	0.3	0.3	0.0	0.9	0.7	0.1
	enhanced cover 2003	0.0	0.0	0.1	0.0	0.1	0.2
	enhanced cover 2056	0.0	0.0	0.1	0.0	0.1	0.2
	enhanced cover 2056 late	0.1	0.0	0.1	0.4	0.1	0.2
	enhanced cover 2215	0.0	0.0	0.1	0.0	0.1	0.3
	proposed cover 2056	0.0	0.1	0.1	0.0	0.2	0.2
Rural Resident Child	site soils 2056	1.3	1.0	0.2	1.8	1.4	0.2
	enhanced cover 2003	0.0	0.1	0.3	0.0	0.1	0.4
	enhanced cover 2056	0.0	0.1	0.3	0.0	0.1	0.5
	enhanced cover 2056 late	0.6	0.2	0.3	1.0	0.3	0.4
	enhanced cover 2215	0.0	0.1	0.4	0.0	0.1	0.6
	proposed cover 2056	0.0	0.2	0.3	0.0	0.3	0.5
Native American Adult	site soils 2056	0.8	0.6	0.1	2.6	2.0	0.3
	enhanced cover 2003	0.0	0.0	0.2	0.0	0.2	0.6
	enhanced cover 2056	0.0	0.0	0.2	0.0	0.2	0.7
	enhanced cover 2056 late	0.4	0.1	0.2	1.3	0.4	0.6
	enhanced cover 2215	0.0	0.0	0.2	0.0	0.2	0.8
	proposed cover 2056	0.0	0.1	0.2	0.1	0.5	0.7
Native American Child	site soils 2056	2.6	2.0	0.3	4.1	3.2	0.5
	enhanced cover 2003	0.0	0.1	0.6	0.1	0.3	1.0
	enhanced cover 2056	0.0	0.1	0.7	0.1	0.3	1.1
	enhanced cover 2056 late	1.1	0.4	0.6	2.1	0.6	1.0
	enhanced cover 2215	0.0	0.1	0.7	0.1	0.3	1.3
	proposed cover 2056	0.1	0.4	0.7	0.1	0.7	1.1

Table 5.1.3 Groundwater-Related Dose by Scenario and Cover Type

	mrem/y	I-129			U-234		
Scenario	Timeframes	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Rural Resident Adult	site soils 2056	0.1	0.0	2.8	0.9	0.0	0.0
	enhanced cover 2003	0.0	0.0	2.4	0.1	0.2	0.2
	enhanced cover 2056	0.0	0.0	2.5	0.1	0.2	0.2
	enhanced cover 2056 late	0.1	0.0	2.3	1.1	0.1	0.1
	enhanced cover 2215	0.0	0.0	2.9	0.1	0.2	0.2
	proposed cover 2056	0.0	0.0	2.6	0.1	0.2	0.2
Rural Resident Child	site soils 2056	0.1	0.0	3.4	1.1	0.0	0.0
	enhanced cover 2003	0.0	0.0	2.9	0.1	0.2	0.3
	enhanced cover 2056	0.0	0.0	3.1	0.1	0.2	0.3
	enhanced cover 2056 late	0.1	0.0	2.8	1.4	0.1	0.1
	enhanced cover 2215	0.0	0.0	3.6	0.1	0.2	0.3
	proposed cover 2056	0.0	0.0	3.2	0.1	0.2	0.2
Native American Adult	site soils 2056	0.2	0.0	5.4	6.2	0.0	0.0
	enhanced cover 2003	0.0	0.0	4.6	0.4	1.0	1.6
	enhanced cover 2056	0.0	0.0	4.9	0.4	1.0	1.6
	enhanced cover 2056 late	0.2	0.0	4.4	7.5	0.7	0.7
	enhanced cover 2215	0.0	0.0	5.7	0.4	1.0	1.6
	proposed cover 2056	0.0	0.0	5.1	0.4	1.3	1.0
Native American Child	site soils 2056	0.2	0.0	6.0	1.1	0.0	0.0
	enhanced cover 2003	0.0	0.0	5.1	0.1	0.2	0.3
	enhanced cover 2056	0.0	0.0	5.4	0.1	0.2	0.3
	enhanced cover 2056 late	0.2	0.0	4.9	1.4	0.1	0.1
	enhanced cover 2215	0.0	0.0	6.3	0.1	0.2	0.3
	proposed cover 2056	0.0	0.0	5.6	0.1	0.2	0.2

Table 5.1.3 Groundwater-Related Dose by Scenario and Cover Type

	mrem/y	U-235			U-238		
Scenario	Timeframes	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Rural Resident Adult	site soils 2056	0.1	0.0	0.0	4.6	0.0	0.0
	enhanced cover 2003	0.0	0.0	0.0	0.3	0.8	1.2
	enhanced cover 2056	0.0	0.0	0.0	0.3	0.8	1.2
	enhanced cover 2056 late	0.1	0.0	0.0	5.6	0.5	0.5
	enhanced cover 2215	0.0	0.0	0.0	0.3	0.8	1.2
	proposed cover 2056	0.0	0.0	0.0	0.3	1.0	0.8
Rural Resident Child	site soils 2056	0.1	0.0	0.0	5.3	0.0	0.0
	enhanced cover 2003	0.0	0.0	0.0	0.3	0.9	1.4
	enhanced cover 2056	0.0	0.0	0.0	0.3	0.9	1.4
	enhanced cover 2056 late	0.1	0.0	0.0	6.4	0.6	0.6
	enhanced cover 2215	0.0	0.0	0.0	0.3	0.9	1.4
	proposed cover 2056	0.0	0.0	0.0	0.4	1.1	0.9
Native American Adult	site soils 2056	0.6	0.0	0.0	30	0.0	0.0
	enhanced cover 2003	0.0	0.1	0.2	1.9	5.0	7.9
	enhanced cover 2056	0.0	0.1	0.2	1.9	5.0	7.9
	enhanced cover 2056 late	0.8	0.1	0.1	36	3.4	3.2
	enhanced cover 2215	0.0	0.1	0.2	1.9	5.0	7.9
	proposed cover 2056	0.0	0.1	0.1	2.1	6.1	4.9
Native American Child	site soils 2056	0.2	0.0	0.0	7.8	0.0	0.0
	enhanced cover 2003	0.0	0.0	0.0	0.5	1.3	2.1
	enhanced cover 2056	0.0	0.0	0.0	0.5	1.3	2.1
	enhanced cover 2056 late	0.2	0.0	0.0	9.4	0.9	0.8
	enhanced cover 2215	0.0	0.0	0.0	0.5	1.3	2.1
	proposed cover 2056	0.0	0.0	0.0	0.6	1.6	1.3

Table 5.1.3 Groundwater-Related Dose by Scenario and Cover Type

	mrem/y	Pu-238			Pu-239		
Scenario	Timeframes	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Rural Resident Adult	site soils 2056	0.6	0.0	0.0	0.8	0.0	0.0
	enhanced cover 2003	0.2	0.0	0.0	0.1	0.2	0.2
	enhanced cover 2056	0.2	0.0	0.0	0.1	0.2	0.2
	enhanced cover 2056 late	1.3	0.0	0.0	1.3	0.1	0.1
	enhanced cover 2215	0.2	0.0	0.0	0.1	0.2	0.2
	proposed cover 2056	0.2	0.0	0.0	0.2	0.1	0.1
Rural Resident Child	site soils 2056	0.6	0.0	0.0	0.7	0.0	0.0
	enhanced cover 2003	0.2	0.0	0.0	0.1	0.1	0.2
	enhanced cover 2056	0.2	0.0	0.0	0.1	0.1	0.2
	enhanced cover 2056 late	1.1	0.0	0.0	1.0	0.1	0.0
	enhanced cover 2215	0.2	0.0	0.0	0.1	0.1	0.2
	proposed cover 2056	0.2	0.0	0.0	0.1	0.1	0.1
Native American Adult	site soils 2056	14	0.0	0.0	17	0.0	0.0
	enhanced cover 2003	4.5	0.0	0.0	3.1	3.4	4.2
	enhanced cover 2056	4.5	0.0	0.0	3.1	3.4	4.2
	enhanced cover 2056 late	28	0.0	0.0	27	1.4	1.2
	enhanced cover 2215	4.5	0.0	0.0	3.1	3.4	4.2
	proposed cover 2056	4.5	0.1	0.0	3.2	2.8	1.8
Native American Child	site soils 2056	1.5	0.0	0.0	1.9	0.0	0.0
	enhanced cover 2003	0.5	0.0	0.0	0.3	0.4	0.5
	enhanced cover 2056	0.5	0.0	0.0	0.3	0.4	0.5
	enhanced cover 2056 late	3.1	0.0	0.0	2.9	0.2	0.1
	enhanced cover 2215	0.5	0.0	0.0	0.3	0.4	0.5
	proposed cover 2056	0.5	0.0	0.0	0.3	0.3	0.2

Table 5.1.3 Groundwater-Related Dose by Scenario and Cover Type

	mrem/y	H-3			Combined All Radionuclides Offsite		
Scenario	Timeframes	0-500	500-1000	>1000y	0-500	500-1000	>1000y
Rural Resident Adult	site soils 2056	7.6	0.0	0.0	16.1	1.0	4.4
	enhanced cover 2003	6.8	0.0	0.0	7.5	1.2	5.2
	enhanced cover 2056	6.8	0.0	0.0	7.5	1.2	5.5
	enhanced cover 2056 late	25	0.0	0.0	36	0.9	4.1
	enhanced cover 2215	6.8	0.0	0.0	7.5	1.2	6.6
	proposed cover 2056	6.8	0.0	0.0	7.6	1.6	5.0
Rural Resident Child	site soils 2056	8.0	0.0	0.0	19	2.5	5.5
	enhanced cover 2003	7.2	0.0	0.0	7.9	1.5	6.4
	enhanced cover 2056	7.2	0.0	0.0	7.9	1.5	6.9
	enhanced cover 2056 late	27	0.0	0.0	39	1.3	5.2
	enhanced cover 2215	7.2	0.0	0.0	7.9	1.5	8.2
	proposed cover 2056	7.2	0.0	0.0	8.0	2.1	6.3
Native American Adult	site soils 2056	8.5	0.0	0.0	80	2.7	8.3
	enhanced cover 2003	7.6	0.0	0.0	17.6	9.8	21
	enhanced cover 2056	7.6	0.0	0.0	17.6	9.8	21
	enhanced cover 2056 late	29	0.0	0.0	129	6.2	12
	enhanced cover 2215	7.6	0.0	0.0	17.6	9.8	23
	proposed cover 2056	7.6	0	0.0	18.0	11	15
Native American Child	site soils 2056	8.1	0.0	0.0	28	5.3	8.5
	enhanced cover 2003	7.2	0.0	0.0	8.7	2.4	10
	enhanced cover 2056	7.2	0.0	0.0	8.7	2.4	11
	enhanced cover 2056 late	27	0.0	0.0	47	2.2	8.5
	enhanced cover 2215	7.2	0.0	0.0	8.7	2.4	13.0
	proposed cover 2056	7.2	0.0	0.0	8.9	3.4	10

5.2 Summary of Results

Tables 5.1.1 and 5.1.2 summarize the dose and risk, respectively, for all scenarios. Table 5.1.3 summarizes the groundwater related dose for all scenarios. The primary source for the offsite dose is from the groundwater, with only a minor contribution from radon. Of all the alternatives analyzed, the Enhanced Late GeoSynthetic/GCL Cover stands out as providing the least protection from dose. Resulting doses for this cover range from 36 mrem/year for the Rural Resident Adult, to 130 mrem/year for the Native American Adult.

The impact of operating the site until 2056 or until the entire site is filled (estimated as 2215) appears to have little impact on the final estimates of dose. As previously discussed, the predicted groundwater concentrations are driven by the 40 years of uncovered trenches and corresponding high infiltration rates. The end result is that the predicted groundwater concentrations remain almost the same for the various time frames (for the same type of cover). It was more conservative to model the intruder accessing the portion of the site that contains the original waste, as it contained comparatively a greater concentration of contaminants. For radium 226, the current practice of segregating the waste (placing high-activity waste at the bottom of the trenches) serves to further reduce the potential impact of the radon for even large amounts of future disposed radium 226.

The US Ecology Proposed cover provides the lowest predicted offsite results at 18 mrem/y to the Native American Adult. As mentioned in the discussion, the lower peak dose for this cover is due to the greater infiltration rate over a longer period of time. The enhanced covers, in comparison, have a significantly lower infiltration rate while the covers remain intact, but result in a contaminant flux peak after cover failure. Since a single value is used for this portion of the analysis, the net result is a higher predicted dose for the enhanced covers.

Differences in the dose and risk estimates when comparing the Native American results to the rural residential results, aside from the large contributions from the sweat lodge, can be attributed to a number of factors; namely:

- Enhanced contribution as a result of an assumed increased consumption of fruits and vegetables, as well as a significantly greater assumed fraction grown locally (62.5% grown locally for the Native American, versus 30%-40% for the rural resident)
- Increased consumption of water to account for the additional water loss while using the sweat lodge
- Slight differences in the amount of meats and milk consumed, as compared to the rural resident, and a greater assumed contaminant concentration for the organ meats

Most of the differences between the rural resident and the Native American scenarios can be attributed to differences in habits and consumption patterns between the two.

Several of the differences can simply be attributed to modeling assumptions (greater percentage locally grown, greater contaminant concentration in organ meats) that may or may not reflect actual exposure conditions.

6.0 Radiological Risk Uncertainty Analysis

The radiological dose analysis for the FEIS presents single-point estimates of dose and risk for closure of the commercial LLRW disposal site. While reported dose values may be high for the single-point estimates, the uncertainty for these estimates is several orders of magnitude, as will be shown in this analysis. Estimates of dose from exposure to radiation for future events are generally recognized to have high uncertainty. This uncertainty, combined with uncertainties associated with the prediction of contaminant movement in the groundwater, and habits and lifestyles of individuals thousands of years in the future, make the overall uncertainty even higher. For the single-point estimates of dose, conservative input values were intentionally used.

The purpose of this uncertainty analysis is to provide individuals with an estimate of the potential exposures in the future, and to take into consideration the likelihood of a rural resident (subsistence) scenario. This realism is included in the uncertainty analysis by taking into consideration the possible range of a given parameter such as the drinking water intake rate, amount of food grown, time spent on the contaminated land, etc. Information available for parameters is reviewed, and a distribution of potential results is derived and included. Once all of these parameters are taken into consideration, the overall dose and risk model is run, using a Monte Carlo approach. This approach allows each parameter specified to vary within a predicted distribution in order to determine the most likely dose to an individual, as well as the upper bound of doses. The list of parameters chosen for the uncertainty analysis is included in Attachment 1, the Uncertainty Parameters Table.

The uncertainty analysis has been divided into five steps:
1. Source Term Uncertainty
2. Groundwater Uncertainty
3. Uncertainties Associated with Exposure Parameters
4. Radiation Dosimetry Uncertainty
5. Uncertainties Associated with Risk Projection Models

Groundwater uncertainty, in addition to the brief discussion below, is included with the Groundwater Pathway Analysis in Appendix 3. Uncertainties associated with exposure parameters are considered in three general divisions. The first division is physiological parameters such as body weight and inhalation rate. The second division is behavioral factors such as the drinking water rate, time spent indoors, etc. The third division is environmental factors such as plant uptake rates, radon diffusion rates, etc. Radiation dosimetry uncertainty includes a wide application of probable uncertainty. The uncertainty is limited to individual differences related to organ size, uptake, and

retention. Other uncertainties are qualitatively addressed. Finally, the estimated uncertainty associated with risk is discussed and quantified.

It should be pointed out that, due to the fact that these exposures will occur in the future, there is no way to validate the model used to estimate the results. One must make the assumption that the mathematical relationships developed to represent contaminant transport and exposure accurately mimic actual exposure conditions and contaminant transport through the environment. The uncertainty analysis is therefore limited to determining the range of possible results, given likely variations for numerous input parameters.

6.0.1 The Focus of the Uncertainty Analysis

The results presented in the EIS are based upon a single-point estimate for a number of scenarios. The input parameters used in the scenarios are intended to serve the following purpose:

- For the rural resident scenario, the dose and risk estimates are designed to be sufficiently protective of the general population through the use of a rural setting. The dose results are intended to estimate the 95 percentile.
- For the Native American scenario, the dose and risk estimates are intended to represent the average member of this critical group.
- For the child scenarios, the results are intended to represent the endpoints used in the corresponding adult scenarios.

For these scenarios, however, one cannot adequately determine whether the target dose goals are met without the use of an uncertainty analysis for the input parameters. Limited data exist to assess the uncertainty of the Native American scenario. An uncertainty analysis for the Native American scenario is therefore not performed. Sufficient information is available for the rural resident scenario (general population) to arrive at an overall uncertainty estimate.

The uncertainty analysis for the Rural Resident Adult includes a number of parameters that allow for an estimate of the likelihood of an individual of the general population to live in a rural setting. The two key parameters that allow for the inclusion of likelihood of this information are the locally grown food consumption rates and the hours spent indoors and outdoors. These data are available in the most recent version of the EPA Exposure Factors Handbook [U.S. EPA, 1997].

The Monte Carlo analysis [Decisioneering, 1996] is used to determine the uncertainty surrounding the single-point estimates for the rural resident scenario. The inputs for the Monte Carlo analysis are the probability distributions for key parameters. The distributions used in this analysis are considered subjective, as they are based on the most current information that will be subject to change as more information becomes available in the future.

The sensitivity analysis for this model is performed by Crystal Ball [Decisioneering, 1996] and estimates the sensitivity by calculating rank correlation coefficients between all of the input parameters and the end result (the dose or risk). The modeler must first make a few assumptions about what parameters are likely to be an important contribution to the final results, prior to conducting the first sensitivity analysis run. This information is obtained from other environmental studies performed in recent years [U.S. DOE, 1996; U.S. DOE, 1998; and NCRP, 1999].

The shape of the probability distributions reflects the depth of information available for a given parameter [NCRP, 1996]. For parameters such as the weathering constant, sufficient data exist to estimate the range and likely value, but insufficient information exists to further define the distribution. The weathering constant is therefore assigned a triangular probability distribution. Greater information exists on the drinking water (tap water) intake rate for the general population and allows for further definition of the distribution as log-normally distributed, with estimated percentiles on the distribution. In some instances, parameters are assigned a triangular distribution due to their minor impact on the overall dose estimate. The triangular distribution for the irrigation rate is a good example of an area where increased research or modifying data on the overall range and distribution would not affect the overall results.

6.0.2 Segregation of Uncertainty and Variability

In uncertainty analyses, two types or sources of variation exist: uncertainty and variability [Decisioneering, 1998]. Parameters exhibit uncertainty, generally due to insufficient information about the true value (or range of values). The wet-to-dry conversion factor for plants is an example of a parameter with some uncertainty. Each plant of interest has a different moisture content. If one is able to quantify the moisture content of all of the plants consumed, with their appropriate consumption weight, then an accurate means and range can be used.[86] Parameters exhibit variability due to the random fluctuations within a population. Examples include intake estimates of food or water (i.e., no two individuals are exactly the same).

It is also possible for parameters to exhibit both uncertainty and variability. Such parameters are termed second-order random variables. The soil-to-plant concentration factor is an example of a parameter with uncertainty and variability about the true value. The soil-to-plant concentration factor would exhibit some variation when only one plant is of interest. This variation is due to differences in the chemical form of the radionuclide, soil characteristics, distribution of the radionuclide within the soil, and internal contaminate distribution within the plant [Till and Meyer, 1983]. In addition to the individual plant variability, uncertainty also exists due to the many varieties of plants that are grown and consumed.

[86] The individual variability among a given type of vegetable or fruit is assumed to be small, and is therefore neglected.

6.1 Source Term Uncertainty

A majority of the I-129 and Tc-99 disposed at the commercial LLRW disposal site is commercial reactor waste. The quantity of Tc-99 and I-129 reported on disposal manifests is based upon scaling factors. In actual practice, the minimum detectable activity (MDA) of I-129 and Tc-99 was used for the calculation of the scaling factor, and resulted in overestimates of the actual quantities of Tc-99 and I-129 by anywhere from 100 to 10,000 [U.S. NRC, 1994]. The overestimate resulted from the use of an upper bound (the MDA), as opposed to determining the actual concentration in the waste or by utilizing a more accurate scaling factor. A more accurate method for determining the disposal quantities of Tc-99 and I-129 has been developed by Vance and Associates [U.S. NRC, 1994; Vance, 1998]. The improved methodology, if applied, would reduce the over-conservatism to within a factor of 10 (as opposed to the current range of 100 to 10,000). It is very likely that if the source terms for Tc-99 and I-129 were accurately modeled, very little I-129 or Tc-99 would be predicted. For this uncertainty analysis, the potential uncertainty in the Tc-99 and I-129 source term is not considered. In the FEIS, however, uranium and plutonium tend to dominate the dose contributions, making the impact from either iodine or technetium small.

Significant effort has been spent by DOH staff and US Ecology staff (since the DEIS), auditing and verifying the uranium source term for the LLRW facility. The estimated uranium 235 and 238 activities are now believed to be accurately reported.

6.2 Groundwater Uncertainty

The uncertainty analysis for the groundwater modeling provided the output in terms of predicted groundwater concentrations for a number of timeframes from 0 to 1,000 years. In this uncertainty analysis, the three peak time periods were analyzed, as they represent the upper bound values for exposure. The groundwater output for 60 years, 1,000 years, and 10,000 years is 500 realizations for each radionuclide. The resulting radiological uncertainty analysis incorporated these groundwater realizations by randomly selecting among the realizations, while maintaining the correlation among all the radionuclides for a given timeframe of interest. See the Groundwater Pathway Analysis for the uncertainty analysis discussion related to the groundwater portion.

6.3 Uncertainties Associated with Human Exposure Assessment

This section includes a review of some of the parameters influencing the dose or risk. The distributions and references for all of the parameters are located in Attachment 1.

Consumption Rate

Information on the consumption of vegetables, fruits, dairy products, meats, and eggs is summarized in the EPA Exposure Factors Handbook [U.S. EPA, 1997]. The data provided in Chapter 13 of Volume II for western states are specifically applied, as this

directly relates to the consumption of homegrown products. As the rural resident is assumed to be a member of the overall population, the consumption distribution has the fraction of the overall population consumption applied, in order to truly represent the population as a whole. A limitation of these data is that the reported values are provided as g/kg-day, as the intake rates are indexed to the body weights of the individuals in the survey. The survey included adults and children. The g/kg-day ingestion values are multiplied by the assumed 70-kg adult weight in order to arrive at a consumption (g/day) rate basis used throughout the EIS calculations. The log-normally-distributed data's 5% and 95% values are provided in Table 6.1.

Table 6.1 Consumption Rates for Food Products (g/day)

Food Type	5%	95%
Fruit	4	600
Leafy Vegetables	0.25	63
Non Leafy Vegetables	1	290
Beef	1.1	131
Poultry	0.9	106
Dairy	12.6 ml/d	2000 ml/d
Eggs	14.4	95.2

Some simplifying assumptions made in the conversions:

- Milk is assumed to be the total dairy consumption. The density of milk is assumed the same as water.
- Data from the *Exposure Factors Handbook* are only available for total meat for consumers only. These data are applied to beef and poultry by using NUREG 5512. Table 13-8 of the *Exposure Factors Handbook* is used to obtain those fractions.
- The *Exposure Factors Handbook* provides combined data for total vegetables. These data are then applied to leafy and non-leafy vegetables by assuming the fractions of consumption provided in NUREG 5512 (17.8% for leafy vegetable intake, 82.2% for non-leafy vegetable intake).

Drinking Water Intake

The range and distribution provided by the EPA *Exposure Factors Handbook* are provided from fitted distributions from Roseberry and Burnmaster. The 5% value is 0.5 l/d; the 95% is 2.5 l/d. Not included in this distribution is the consideration of increased drinking water in a temperate climate. Elevated temperatures exist in the Hanford area for about three to four months of the summer and may affect the distribution, although this possibility has not been explicitly analyzed. The 3-l/d drinking water value used in the radiological analysis for this EIS is approximately 97.5% value for this distribution. For the model, the intake frequency is assumed to be 365 d/y, as the intake rate is adjusted for frequency.

Distribution Coefficient – Tc-99 and Cl-36

The distribution coefficient information is obtained from Appendix E of the *Composite Analysis* [Kincaid, et al, 1998]. For the dry disposal site, the estimated range of the distribution coefficient extends from -2.8 to 0.6, with a most likely value of 0. The negative value for the distribution coefficient cannot be completely modeled without the resulting infiltration rates estimates becoming negative as well.[87] The resulting range is truncated with a lower bound of -0.07 and an upper bound of 0.6. The distribution of the distribution coefficient is a step-wise distribution, with a mode of 0 and an exponential decay to 0.6 [Fayer, 1999].[88]

Soil-to-Plant Concentration Factors of Tc-99 for Leafy Vegetables and Grasses

Information on the 5% and 95% values for both grasses and leafy vegetables is obtained from the International Atomic Energy Agency/International Union of Radioecologists [IAEA, 1994]. The upper bound on the concentration factor is limited by the amount of contaminant available for uptake; i.e., it is possible to model a concentration factor that results in a greater amount of contamination removed from the soil than is deposited in the soil from irrigation. As a result, the upper bound value is limited to the total contamination deposited in a season.[89]

- For leafy vegetables, the geometric mean is taken as 210; the geometric standard deviation (GSD) is 1.5. The upper bound on the log-normally-distributed parameter is 430.
- For grasses, the geometric mean is taken as 210; the geometric standard deviation (GSD) is 2.3. The upper bound on the log-normally-distributed parameter is 680. The upper bound value for grasses is higher than the calculated mass limited value for leafy vegetables, due to the lower estimated plant yield for grasses, as compared to leafy vegetables (i.e., a smaller amount of potential contaminate removal).

Wet-to-Dry Conversion Factors

The EPA *Exposure Factors Handbook* provided information on the moisture content, as well as a table for consumption rates of the various food products, that allowed the weighting of the results to obtain an overall weighted mean value. A triangular distribution was used, with the range being the highest and lowest reported values.

[87] Negative Kd values are possible, as the scale is in relation to the speed of water moving in a soil column. The negative charge of Cl-36 and Tc-99 has the effect of repelling the ions from the surface of the soil particles. This can cause the ions to remain in the larger soil pores, causing them to move down preferential pathways, and in a sense, travel faster than water [Napier, 1999b].

[88] NOTE: The distribution coefficient and any other groundwater-related parameters for this uncertainty discussion are only assumed to apply to the contaminated groundwater that is applied to the food products and used for drinking water. The distribution coefficient values mentioned here affect groundwater movement only after the groundwater has been contaminated. In short, this is a non-recycling model.

[89] For Cl-36 and Tc-99, contaminant transport is sufficiently fast to result in removal of the contaminant prior to the next growing season.

- For leafy vegetables, the weighted mean moisture content is 0.93, with a range of 0.86 to 0.95
- For non-leafy vegetables, the weighted mean moisture content is 0.90, with a range of 0.59 to 0.96
- For fruits, the weighted mean moisture content is 0.80 with a range of 0.74 to 0.92

6.3.1 Critical Parameters for the External Dose Pathway

The estimates of the dose to the intruder from external sources of radiation contain a significant amount of uncertainty. For the uncertainty analysis, the following potential sources of uncertainty or variability are identified:

1. There is a variation of dose due to gender, as compared to calculated. The error is assumed to be uniform, with a ±10% error. The magnitude of the estimate is based upon comparisons of adult sex-specific and hermaphrodite phantoms [Eckerman and Ryman, 1993]. Not included in this estimate is variation due to physical size, as this analysis is for an adult. NCRP Commentary #15 [NCRP, 1998] states that the dose to a baby is perhaps 20% higher than that received by an adult (primarily due to height). It is interesting to note that the corrections are not much different for children as compared to adults [NCRP, 1999].
2. The ratio of the effective dose as compared to the air kerma is about 80% for rotational exposures [NCRP, 1999]. This value is almost independent of energy.
3. There is uncertainty, due to Effective Dose versus Effective Dose Equivalent. FR #12 uses ICRP 26 tissue weighting factors. The fact that the older tissue weighting factors (ICRP 26) are used, as opposed to ICRP 60 recommended values, introduces an error of less than 10% [NCRP, 1999].
4. Variations in the estimate of the exposure time are also large. These include errors in the time spent outdoors (in the contaminated area), as well as time spent indoors. The uncertainty analysis is based upon the data from the EPA *Exposure Factors Handbook.*

6.3.2 Critical Parameters in the Radon Pathway

Radon risk estimates are seldom performed by calculating the dose from an exposure, and then converting directly to risk. Instead, epidemiological data from miners are used to determine the actual risk from exposure. To provide an estimate of the dose received, the risk estimate is converted back to a dose. One salient issue when converting from risk to dose is the appropriate conversion factor to use. Radon and its progeny predominately affect only the lung. The inclusion of non-fatal contributions, and relative length of life lost, only reduces the fatality probability by about 5% [ICRP, 1990]. Additional detriments to other tissues of the body from radon exposure only increase the risk by about 2% [ICRP, 1993]. These differences between the fatal coefficient and the overall detriment are small enough to allow the use of the fatality coefficient for an overall measure of detriment. So, the risks are essentially the same for radon exposure, whether one chooses an overall health detriment or simply a fatal cancer coefficient.

The uncertainty for the radon estimates is as large as those for the groundwater portion of the analysis. Attachment 1 provides the results of the radon-related analysis modeled for uncertainty. Some specific sources of uncertainty are discussed below:

- The radon emanation coefficient would be expected to vary from about 0.14 to 0.28, depending upon the soil type [Yu, et al, 1993]. This range of values is somewhat misleading for the LLRW facility, as up to 80% of the radium source term is in the form of discrete sealed sources encased in concrete. Such a sealed source would not be expected to have a significant emanation fraction for perhaps several thousand years. The effect of sealed sources after 500 years is not considered in the uncertainty analysis and will result in a high bias of results. The effective diffusion coefficient is dependent upon the type of soil, porosity, and percent moisture. The radon diffusion calculations relied upon Nuclear Regulatory Guidance 3.64. This guidance, as expected, is somewhat conservative. Other sources of models for the calculation of the radon flux differ by as much as 50% lower than the values used [Hart, et al, 1986]. This potential high bias due to the model is not considered in the analysis.
- Another source of uncertainty is the effective dose per unit exposure factor. Whether this value is derived based upon the energy deposited in the lungs or based upon the epidemiology of the miner studies, numerous uncertainties exist. For the lung, uncertainties exist as to the target cells of interest and the location. Uncertainties inherent in epidemiological modeling include lack of statistical size, adequate control groups, extrapolation from miners to home exposure conditions, adequate control for competing causes of cancer, etc. The range used for modeling is based upon the information provided by the EPA for their proposed drinking water rule [National Research Council, 1999].
- Estimates of the hours of occupancy indoors available in the literature range from about 50% to 100%. For this analysis, the data for the time spent indoors and outdoors are based upon the EPA *Exposure Factors Handbook*.
- The discrete fraction of radium disposed is a primary driver for the estimated radon contribution. In the deterministic analysis, it is assumed that the substantial barriers surrounding the sealed radium sources are degraded to such an extent that the sealed sources contribute to the radon flux by 500 years. Based upon DOH staff review of the integrity of the PGE reactor vessel and related components, it seems clear that the stainless steel and/or lead surrounding the sealed sources and further encased within a drum of concrete would withstand degradation for substantially longer than 500 years, but certainly not as long as a solid stainless steel reactor vessel. For the uncertainty analysis, the discrete fraction was assumed to remain intact for several thousand years, at which point the contribution to dose from the increased emanation would be offset by the decay of the source.

6.4 Uncertainty Associated with Radiation Dosimetry

The EPA *Radiation Exposure and Risks Assessment Manual* (RERAM) [U.S. EPA, 1996] provides a comprehensive list of the sources of uncertainty in radiation dosimetry. The uncertainties are due to the model itself (as a simulation of actual processes within

a human body) and parameter variability caused by variation among individuals or measurement error. The sources of uncertainty listed by the EPA include (verbatim):

- Uncertainty in the formulation of the mathematical models for
  -deposition of activity in the lung and translocation of inhaled activity into the blood,
  -translocation and absorption of ingested activity into the blood,
  -distribution and retention of activity from blood to various systemic organs and tissues, and
  -calculation of the absorbed dose to an organ or tissue from activity in that and other organs and tissues;

- Uncertainty in the model parameters, including:
  -parameters in the biokinetic and dose models (e.g., GI absorption fraction, lung clearance class, organ deposition fractions and retention times, organ masses and geometries, etc.), and
  -anatomical and physiological data for characterizing the population of interest.

Dunning and Schwarz [Dunning and Schwarz, 1981] evaluated the uncertainty of estimates of dose to the thyroid from I-131, due to the variability of thyroid mass, uptake and retention of ingested iodine. Using Monte Carlo methods, they determined that the resulting frequency distributions are highly skewed log-normally-distributed, with a geometric standard deviation (GSD) of 1.8. Napier [U.S. DOE, 1998] interpreted these data for application to the uncertainty of all dose conversion factors and rounded the GSD to 2.[90] NCRP 129 [NCRP, 1999] evaluated available data for both inhalation and ingestion dose conversion factors (DCF) and found that the GSD ranged from 1.4 to 2.2 for inhalation conversion factors. The ingestion DCF uncertainty ranged from a GSD of 1.25 to 2.5, depending upon the radionuclide. Although this EIS analysis did not differentiate the uncertainty based upon pathway and radionuclide, a GSD of 2.0 for all radionuclides and pathways is viewed as sufficiently representative.

6.5 Uncertainty Associated with Risk Projection Models

The risk uncertainty analysis was performed in the DEIS and was not repeated for the FEIS. Please refer to the DEIS and more specifically NCRP 126 for more information related to risk uncertainty.

6.6 Results

The uncertainty analysis solely focuses on the Enhanced Composite GCL cover for 2056. The discussions below are segregated into the three time periods of interest: 60 years, 1,000 years, and 10,000 years. Figure 25 of the Groundwater Report provides a graphical output of the drinking water dose, assuming 2 l/d ingestion rates. Although in

[90] The GSD matches closely with the information recently published in NCRP 129, which recommends a GSD of 2.2 for most radionuclides.

the radiological analysis there are significantly more pathways considered, the graph does provide the peak doses and overall uncertainty as time progresses.

6.6.1 Estimated Dose Distributions at 60 Years Post-Closure

One of the basic assumptions for the site is that institutional controls will remain active for at least 100 years post-closure; only the results for the offsite rural resident adult are displayed.

Figure 6.6.1 is a frequency distribution of the results from the 60-year timeframe, the time location of the peak dose. The figure shows the expected dose on the X-axis versus the probability for a given dose on the Y-axis. The dose range extends from 0 to 10 mrem, with a most likely value (the mode) about 2.5 mrem/y, and a 95 percentile upper bound value of 9 mrem/y. Other statistics for the offsite distribution are:

- Mode ≅ 2.5 mrem/y
- Median = 4 mrem/y
- Mean = 4 mrem/y

Figure 6.6.1 Rural Resident Offsite Dose at 60 Years

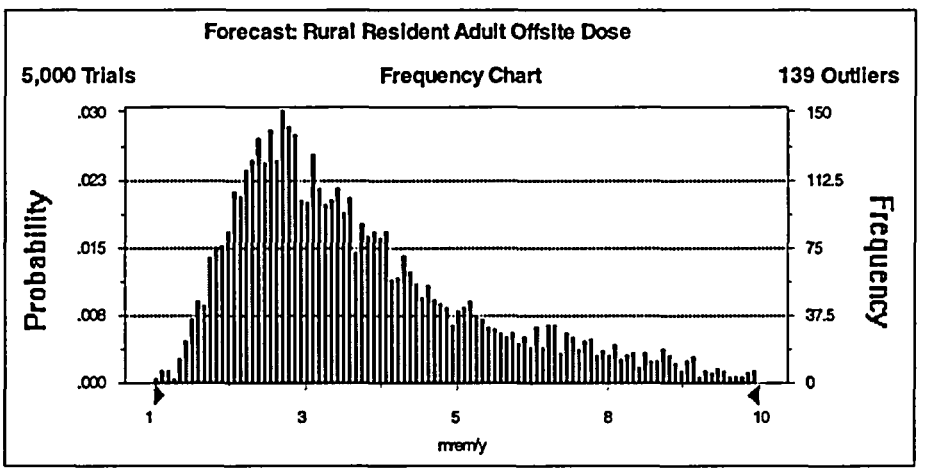
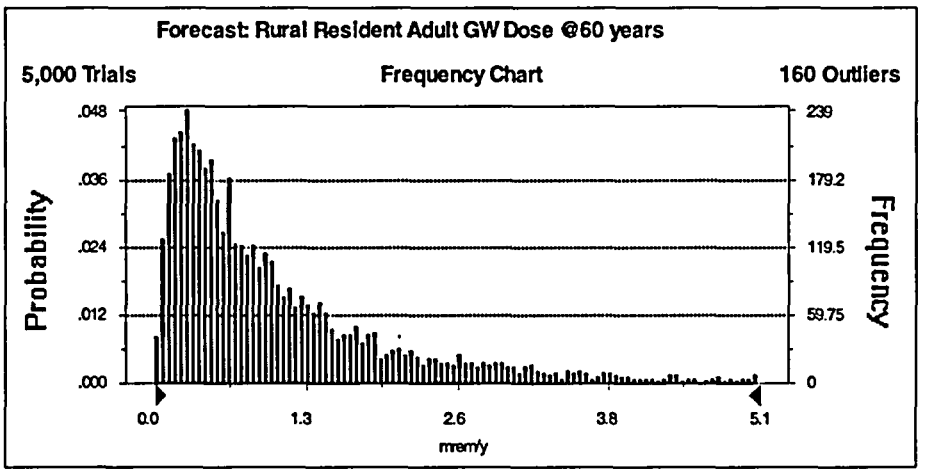


Figure 6.6.2 is a frequency distribution chart showing the groundwater contribution for all radionuclides, with the exception of tritium. The mean, median, and modal values for all groundwater dose contributors, excluding tritium, are all less than 2 mrem/y. The difference between Figures 6.6.1 and 6.6.2 is solely due to the contribution from tritium. In comparison, the Rural Resident Adult's predicted single-point dose is 8 mrem/y from all sources. The single-point estimate is commensurate with the predicted 95% value of 9 mrem/y. Both estimates are well less than the 25-mrem/y offsite limit. In the uncertainty analysis, however, the tritium groundwater concentrations are viewed as conservative, as they do not match the current groundwater concentrations observed under the LLRW facility. In order to limit this conservatism for the groundwater estimates, the uncertainty analysis applied a 3.6 reduction factor to the tritium estimates in order to correct the predicted water concentration for 2000y, to the actual for the

same time period (9900 pCi/l divided by 2,750 pCi/l).[91] This correction factor was applied to all tritium estimates, as the error is assumed constant.[92] Since actual groundwater concentration data are available for the site, it is appropriate to correct predicted results with actual results. Little contribution to dose is observed from other sources such as radon emanation from onsite.

Figure 6.6.2 Rural Resident Groundwater Related Dose (Without Tritium) at 60 Years



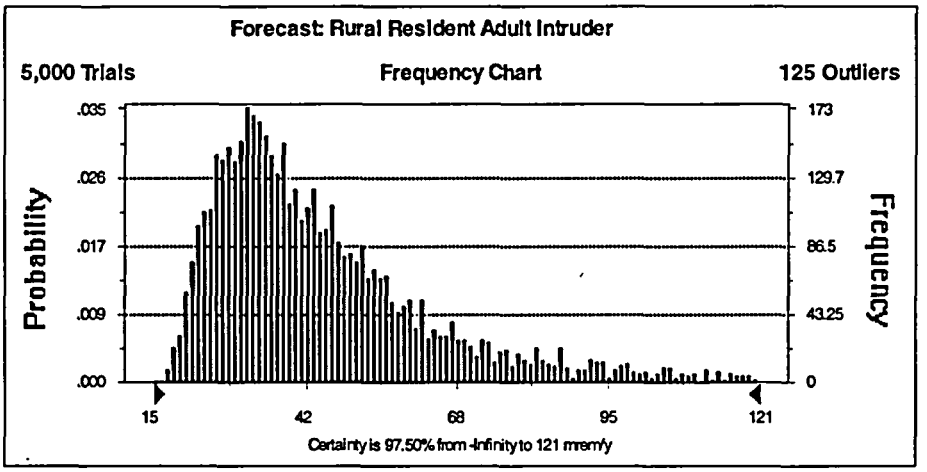
6.6.2 Estimated Dose Distributions at 1000 Years Post-Closure

Figure 6.6.3 displays the results of the dose distribution for the Rural Resident Adult Intruder at 1,000 year following closure. The 95-percentile value is 97 mrem/y, which is somewhat lower than the 105 mrem/y estimated from the single-point doses reported for the Enhanced Composite GCL Covers. Other statistics for the onsite distribution are:

- Mode ≅ 28 mrem/y
- Median = 46 mrem/y
- Mean = 48 mrem/y

[91] Actual tritium concentration data obtained from the *Calendar Year 2000 Annual Environmental Monitoring Report for the LLRW Facility.* It is not clear that the contamination measured near the LLRW Facility is due to the LLRW facility and is likely to be due to offsite contributions from the Hanford site.
[92] Other factors that make this correction more justified is that even a shift (to a later peak tritium concentration) in the concentration assuming a constant rate increase would result in a significant reduction in the groundwater concentrations due to decay alone.

Figure 6.6.3 **Rural Resident Intruder Dose at 1,000 Years**

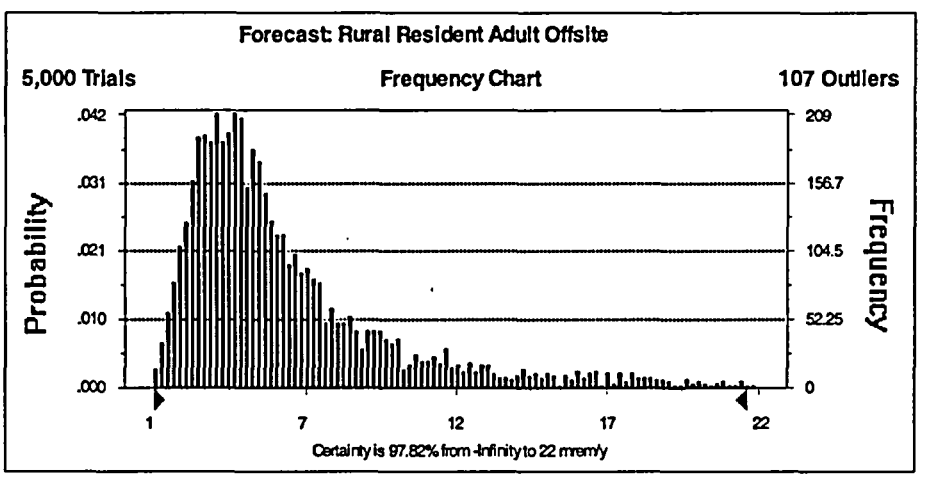


All of the figures for the uncertainty analysis are log-normally distributed and are positively skewed to the right. This distribution graphically reinforces the limited probability that the upper bound estimates represent a likely exposure event.

Figure 6.6.4 displays the results of the dose distribution for the Rural Resident Adult in an offsite setting. The 95-percentile value is 17 mrem/y, which is significantly greater than the 2 mrem/y estimated from the single-point doses reported the Enhanced Composite GCL Covers, but less than the 25 mrem/y offsite dose limit. Other statistics for the onsite distribution are:

- Mode ≅ 3.5 mrem/y
- Median = 5 mrem/y
- Mean = 7 mrem/y

Figure 6.6.4 **Rural Resident Offsite Dose at 1,000 Years**

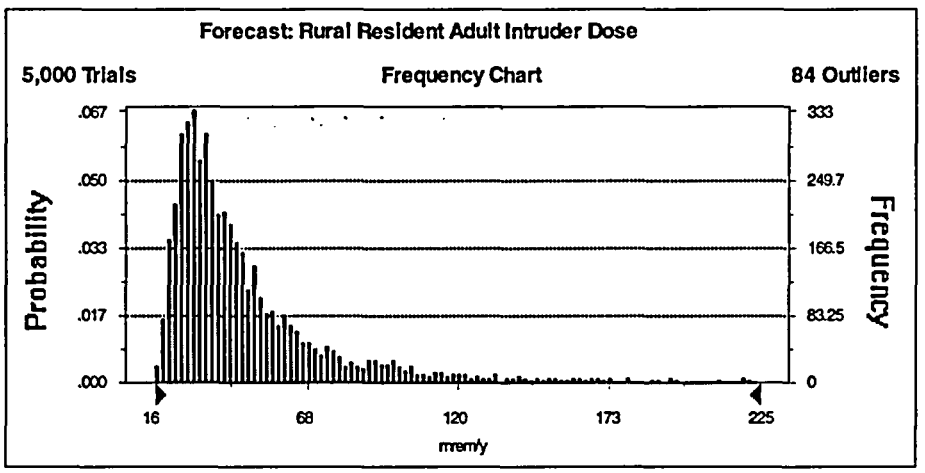


6.6.3 Estimated Dose Distributions at 10,000 Years Post-Closure

Figure 6.6.5 displays the results of the dose distribution for the Rural Resident Adult Intruder. The 95-percentile value is 130 mrem/y, which is somewhat higher than the 93 mrem/y estimated from the single-point doses reported the Enhanced Composite GCL Covers. Other statistics for the onsite distribution are:

- Mode ≅ 30 mrem/y
- Median = 39 mrem/y
- Mean = 54 mrem/y

Figure 6.6.5 Rural Resident Adult Intruder Dose @ 10,000 Years

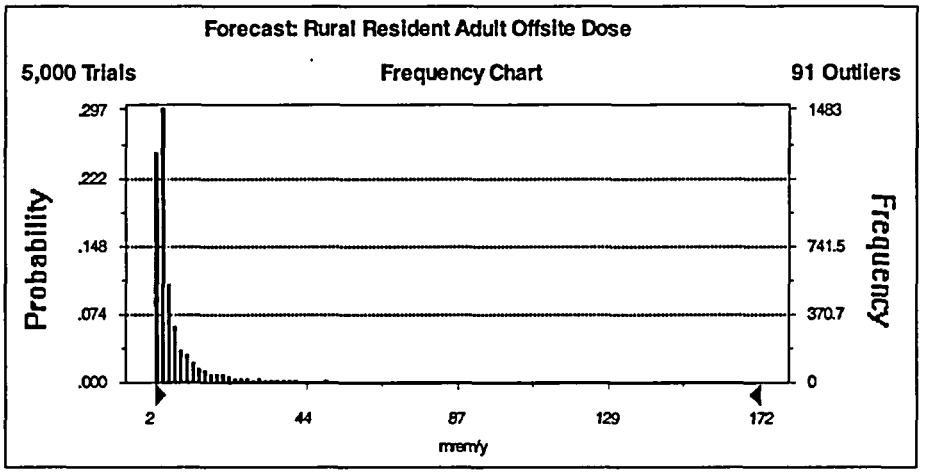


The upper bound value reflects the increased uncertainty associated with projections so far into the future and is further discussed in the Groundwater Appendix. The large variation between an upper bound estimate and the most likely value also indicates the impact of lifestyle assumptions and patterns. Simply put, an intruder who spends most of the day inside the house, consumes a large amount of drinking water every day, and grows a majority of his/her own food, would receive a significantly higher dose than an individual living at the same location who spends a significant amount of time working elsewhere and grows little food locally. This type of variability greatly influences the final results.

Figure 6.6.6 displays the predicted offsite dose to the Rural Resident Adult for the 10,000-year timeframe. The single-point estimate for the adult is 8 mrem/y for the 0 to 500 year timeframe. This single-point estimate is greater than the median estimate of 5 mrem/y but significantly less than the 95% upper bound estimate of 65 mrem/y. All of the contribution to dose for the offsite adult is due to groundwater related exposures, as was alluded to earlier. Other statistics for the offsite estimates are as follows:

- Mode ≅ 4 mrem/y
- Median = 5 mrem/y
- Mean = 18 mrem/y

Figure 6.6.6 Rural Resident Adult Offsite Dose @ 10,000 Years



6.7 CONCLUSIONS

The intent of this uncertainty analysis is to provide an estimate of the overall range and distribution of the dose endpoints. In doing so, evaluating the strength and conservatism of the single-point dose estimates for the rural resident is possible. The results indicate that the offsite single-point estimates are generally less than the 95-percentile values (the intended target endpoint) and are more in line with the median and modal values. The onsite single-point dose estimates appear to be in line or are less than the 95% upper bound estimates. The analyses indicate that the data are a positively skewed log-normal distribution.

Detailed results in Figures 6.6.1 through 6.6.6 only provided results for the peak time periods of 60 years, 1,000 years, and 10,000 years. Those projections are further summarized in Table 6.7.1.

- For the 60-year timeframe, the estimated tritium contribution provides a majority of the predicted offsite dose, with the 95% predicted peak dose of 9.5 mrem/y.
- The 1,000-year offsite dose of 17 mrem/y is also less than the 25 mrem/y limit. The predicted onsite dose to the Rural Resident Adult intruder is essentially the 100-mrem/y onsite limit (97 mrem/y).
- For the 10,000-year timeframe, the offsite dose of 65 mrem/y and the onsite dose of 130 mrem/y are well above their respective limits. Essentially, all of this dose uncertainty can be attributed to the greater uncertainty in the groundwater concentrations. Onsite dose estimates are somewhat misleading for 10,000 years, as the radium contribution would have a significant source term decay[93] that was not accounted for in the uncertainty analysis. This correction would likely reduce the 10,000-year onsite estimates to less than the 100 mrem/y limit.

[93] The current radium 226 disposal site activity is significantly greater than the ingrowth of radium from the uranium source term for the 10,000-year time period.

Table 6.7.1 Rural Resident Adult Summary Uncertainty Results

	Mode	Median	Mean	95%
60-Year Estimates				
Offsite Dose (mrem/y)	2.5	4	4	9.5
Onsite Dose (mrem/y)	NA	NA	NA	NA
1,000-Year Estimates				
Offsite Dose (mrem/y)	3.5	5	7	17
Onsite Dose (mrem/y)	28	46	48	97
10,000-Year Estimates				
Offsite Dose (mrem/y)	4	5	18	65
Onsite Dose (mrem/y)	30	39	54	130

There are a number of factors that are only qualitatively included in the uncertainty analysis. Two in particular are: (1) uncertainties associated with model limitations both in the radon analysis; and, (2) in radiation dosimetry in general. Not including model uncertainty for the radon analysis leads to a high bias in the results. The impact of the radiation dosimetry uncertainties not defined has an unknown impact on the final results.

7.0 RADIOLOGICAL ASSESSMENT CONCLUSIONS

General Statement

This Radiological Risk Assessment has estimated the impact of site closure for a variety of potential covers and closure dates. The results are discussed in terms of expected dose as well as fatal cancer probability. These two expressions of impact, the expected or estimated dose and the corresponding fatal cancer probability, are common methods for expressing the results from radiological exposures. It is also common, however, for chemical risk assessments to express the expected impact in terms of cancer morbidity and mortality, which includes both fatal and non-fatal cancers. In order for the results from both a chemical source and a radiological source to be comparable, the risks units must be equated to the same endpoint.

The radiological results reported in this assessment can be expressed in terms of an overall measure of harm or detriment. This overall measure of detriment includes both fatal and non-fatal cancers, the probability of severe hereditary effects, and the relative length of life lost (due to fatal cancers) [ICRP, 1990]. When taking into consideration all of the additional factors other than the probability of fatal cancers, the risk estimates are increased by approximately 50%.[94] This measure of overall detriment is more comprehensive than that typically used in chemical risk assessments, that include only the probability of fatal and non-fatal cancers. It is important to point out that exposures to some chemicals can have genetic impacts as well (commonly called teratogenic

[94] More specifically, the dose-to-risk conversion factors used in the tables in Chapter 5 would change from 0.0005/Rem to 0.00073/Rem.

agents). Such exposures for chemicals must be estimated on a contaminant-specific basis and may not be included in the reported risk from a chemical exposure.

Considering the potential errors in comparing exposures of radiological and chemical sources and the small estimated chemical contribution from the waste site, the decision was made to report the results from the radiological exposures in terms of the probability of fatal cancer, while providing the method for estimating the overall detriment. Summation of sources of non-radiological exposures (within the 200 areas) with radiological exposures can be performed, but these additions should be carefully reviewed to ensure that the endpoint expressed for each exposure source is the same.

Specific Summary

Included in this analysis is a single-point estimate of the expected dose and risk to an individual, based upon an assumed lifestyle. Due to the large uncertainties in contaminant movement in the groundwater, future land use, and lifestyles of individuals, these single-point estimates are only intended to serve as predictive estimates for the individuals in the scenarios created.

The groundwater concentrations served as the initial basis for a majority of the dose and risk estimates. The subsequent environmental (such as soil to plant transfer factors) and individual parameters (such as time spent indoors, drinking water rates, etc.) were also chosen to provide conservative yet realistic estimates of overall detriment.

The results of the analysis for the onsite and offsite individuals indicate that there are several covers that meet the offsite limit of 25 mrem/y and the onsite limit of 100 mrem/y. By limiting the infiltration and gas emanation, these covers effectively limit the dose received by an individual.

The Proposed Cover, the Asphalt Cover, and the Bentonite Cover all meet the criteria of performing well for both onsite (via the groundwater pathway and gaseous emanation) and offsite (via both the groundwater) scenarios. The Composite GCL Cover meets the offsite limit and only slightly exceeds the onsite limit of 100 mrem/y (at 107 mrem/y).

The Composite GCL analysis for the 2005, 2056, and 2215 closure time periods provides an analysis of the differences that varying the closure date makes. The groundwater analysis indicates clearly that leaving the trenches uncovered has a significant detrimental impact on predicted groundwater contaminant concentrations. This open trench period provides a large initial flux of contaminants that masks most cover and time period differences. While the Composite GCL analysis meets the offsite 25 mrem/y limit for all three time periods, further delays in closing the filled trenches would have an even larger negative impact on future groundwater concentrations and would result in greatly exceeding the 25 mrem/y limit, as is displayed in the Late Enhanced covers results in Table 5.1.1 and Table 5.1.3.

A detailed summary of results in also provided at the conclusion of Section 5. The reader is directed to the summary of Section 5 for further discussions on the impact of various covers and scenarios.

Chapter 6 analyzes the uncertainty for the rural resident adult. The uncertainties included are provided in Attachment 1. Further uncertainties that are only qualitatively included are discussed in the text of this chapter. The results of this analysis show that the single-point estimates of Chapter 5 for offsite and onsite dose and risk estimates are sufficiently conservative for the onsite risk estimates, and are less than the predicted 95% values for the offsite analysis. The uncertainty analysis also shows that the uncertainty in the predicted results increases over time, such that the predicted results in the year 10,000 are subject to a significantly greater potential distribution of results. Perhaps the focal point of the uncertainty analysis and the FEIS radiological analysis in general is that greater emphasis and weight should be given to results in the first 1,000 years, as opposed to results 5,000 to 10,000 years later.

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March 26, 2003

Ms. Nancy Darling
Washington State Department of Health
Division of Radiation Protection
PO Box 47827
Olympia Washington 98504-7827

Dear Ms. Darling:

Enclosed is one copy of two technical reports related to the U.S. Ecology low-level waste disposal site in Hanford Washington. The hardcopy reports are single-sided and unbound to facilitate reproduction if necessary. The first report, *Groundwater Concentrations and Drinking Water Doses with Uncertainty for the U.S. Ecology Low-Level Radioactive Waste Disposal Facility, Richland Washington* address the revised groundwater modeling effort for three cover and closure options. The second report entitled, *FOLAT: A Model for Assessment of Leaching and Transport of Radionuclides in Unsaturated Porous Media,* provides documentation to the unsaturated transport model computer code used in this evaluation.

This work was performed under contract N10996. I have also enclosed three electronic copies of all the files related to this project on a CD. Please review the README.DOC file on the CD for an explanation of their contents.

If you have any questions regarding this work, please feel free to call me at (208) 745-1288

Sincerely

Arthur S. Rood

cc asr files

FINAL REPORT

Groundwater Concentrations and Drinking Water Doses with Uncertainty for the U.S. Ecology Low-Level Radioactive Waste Disposal Facility, Richland Washington

Date March 2003

Submitted to Washington State Department of Health
in partial fulfillment of contract No. N10996

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FINAL REPORT

Groundwater Concentrations and Drinking Water Doses with Uncertainty for the U.S. Ecology Low-Level Radioactive Waste Disposal Facility, Richland Washington

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Arthur S. Rood

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EXECUTIVE SUMMARY

US Ecology Incorporated operates a low-level radioactive waste disposal facility on leased land from the U.S. Department of Energy's Hanford Reservation located near Richland Washington. The Washington State Department of Health (WDOH) is currently developing an Environmental Impact Statement (EIS) for the site. Part of the EIS involves the evaluation of impacts to groundwater of various closure options for the site. A Draft Environmental Impact Statement (DEIS) was completed in 2000 which included an assessment of the groundwater pathway. Recent environmental monitoring data have detected radionuclides in the subsurface below the facility. Because of the way the original transport model was constructed, evaluation of radionuclide concentrations in the unsaturated zone was not possible. Additional information regarding waste disposal history, the effects of open trenches on water infiltration, and evolution of ideas regarding the site conceptual model led WDOH to revisit the groundwater assessment performed for the DEIS.

A new model for radionuclide transport in the unsaturated zone was constructed that incorporated effects of transient infiltration and historical waste disposal rates. Radionuclide inventories were re-evaluated and important radionuclides identified through a two-phase screening approach. Fifteen radionuclides were identified as being important in terms of their potential for groundwater ingestion dose: C-14, Cl-36, H-3, I-129, Pu-238,-239,-240,-242, Ra-226, Tc-99, Th-230, Th-232, U-234, U-235, and U-238. Nickel-63 and Sr-90 were removed from consideration during the screening process, but were retained for model calibration because these radionuclides were detected in measurable quantities in the unsaturated zone beneath trench 5. Assumptions regarding partition coefficients and cover longevity were revisited and modified accordingly.

Radionuclide release rates from the trenches and their transport in the unsaturated zone were calibrated to measured concentrations taken in boreholes beneath trench 5. Measured radionuclide concentration profiles of relatively immobile radionuclides beneath trench 5 could not be explained by dissolved-phase transport, and a colloidal transport model was proposed as an alternative. The colloidal transport model assumes a fraction of the radionuclide inventory (hereafter referred to as the mobile fraction) moves by colloidal transport. We assumed colloidal transport to be represented by a dissolved-phase transport model with no sorption; therefore, radionuclides move with the velocity of water. Calibrated radionuclide mobile fractions ranged from 6.2×10^{-4} to 4.6×10^{-6} for Ni-63, U-238, Sr-90, and Pu-239 and 0.047 for Tc-99. The higher mobile fraction value for Tc-99 reflects its dissolved-phase mobility. For Tc-99, it was necessary to limit the radionuclide release rate from the trenches so that model-predicted radionuclide inventories below the trenches matched inventories extrapolated from the borehole data. Conservative estimates of drinking water dose from the mobile fractions of Ni-63 and Sr-90 were less than 4 mrem yr^{-1}. Therefore, no further evaluation of these radionuclides was warranted beyond model calibration.

The revised transport model also incorporated a cover lifetime of 500 years and partition coefficients that reflect sorption only on the fine material in the rock matrix. Aquifer concentrations and drinking water ingestion doses were calculated as a function of time for five cover design/closure option scenarios. Three cover designs were included in the analysis; a site soils cover which had an infiltration to 20 mm yr^{-1}, an enhanced cover that limited infiltration to

0.5 mm yr^{-1}, and the US Ecology proposed cover that limited infiltration to 2 mm yr^{-1}. Background infiltration was assumed to be 5 mm yr^{-1} after the cover failed.

A parametric uncertainty analysis was performed to evaluate the uncertainty in the model-predicted concentrations and doses. Monte Carlo sampling coupled with simple random sampling was used to propagate uncertainty through the transport model. Uncertainty was not considered for the design-based infiltration rates or the covers, calibrated mobile fractions, exposure scenario parameters (drinking water ingestion rate), or dose conversion factors.

Groundwater concentrations were both higher and lower compared to results in the original DEIS. Higher concentrations were attributed to a) enhanced infiltration through the site during active disposal, b) cover failure after 500 years, and c) uranium solubility. Lower concentrations were attributed to lower leaching rate constants for Tc-99 and Cl-36. Deterministic drinking water doses were dominated by four of the five DEIS radionuclides (I-129, Tc-99, U-235, and U-238) plus H-3, C-14, and the mobile fractions of U-234, U-238, and Pu-239. Total deterministic drinking water doses for the enhanced cover were less than 5 mrem yr^{-1} 100 years after the start of facility operations in the year1965. Doses were1 mrem yr^{-1} between 100 and 1,000 years after 1965, and around 2 mrem yr^{-1} 10,000 years after 1965. The mobile fraction of U-238 dominated the dose 1,000 years after closure, while I-129 and C-14 dominated the doses 10,000 years after closure. Tritium dominated the dose in the 0- to100-year time frame. Doses for the enhanced and proposed cover *while* the cover remained intact were about one order of magnitude lower that those of the site soils cover.

Uncertainty analysis was performed for the enhanced cover only and indicated that the precision of the model-predicted total drinking water dose is roughly a factor of 25 at times less than 100 years after the start of facility operations (1965), and increases to over three orders-of-magnitude for times greater than 100 years. Results may be summarized by the following probability statements:

- We are 95% confident that there is a 95% probability that the *predicted* drinking water dose during the 0–100 year time frame will not exceed 17 mrem yr^{-1}.
- We are 95% confident that there is less than a 5% probability that the *predicted* drinking water doses during the 0–100 year time frame will exceed 17 mrem yr^{-1}.
- We are 95% confident that there is a 95% probability that the *predicted* drinking water dose during the 100–1,000 year time frame will not exceed 4.6 mrem yr^{-1}.
- We are 95% confident that there is less than a 5% probability that the *predicted* drinking water doses during the 100–1,000 year time frame will exceed 4.6 mrem yr^{-1}.
- We are 95% confident that there is a 95% probability that the *predicted* drinking water dose during the 1,000–10,000 year time frame will not exceed 28 mrem yr^{-1}.
- We are 95% confident that there is less than a 5% probability that the *predicted* drinking dose during the 1,000 –10,000 year time frame will not exceed 28 mrem yr^{-1}.Conversely, there is an equal probability (5%) that the dose during the 1,000–10,000 year time frame is less than 4.1×10^{-3} mrem yr^{-1}.

The uncertainty analysis provides a measure of the precision of the transport model and should not be interpreted as the probability of any real or actual exposure occurring. It is simply a measure of the precision by which the model can estimate concentrations and doses far into the future.

Overall, the assessment integrates natural processes that govern the transport radionuclides in the subsurface, with known waste disposal histories, past operational practices, and future closure plans of the site into a transport model that estimates both past and future radionuclide migration from the US Ecology low-level radioactive waste site. Conservative assumptions were made where uncertainty exists and therefore, these results should be viewed as conservative estimates of radionuclide concentrations and drinking water doses.

CONTENTS

FIGURES

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INTRODUCTION

US Ecology Incorporated operates a low-level radioactive waste disposal facility on leased land from the U.S. Department of Energy's Hanford Reservation located near Richland Washington. The Washington State Department of Health (WDOH) is currently developing an Environmental Impact Statement (EIS) for the site. Part of the EIS involves the evaluation of impacts to groundwater of various closure options for the site. A Draft Environmental Impact Statement (DEIS) was completed in 2000. The groundwater pathway analysis was documented in Dunkelman (2000). Radionuclide transport in the unsaturated zone and aquifer was evaluated in a separate document (Rood 2000a) which was later integrated with the DEIS. Groundwater concentrations were calculated for a single cover design and aquifer concentration included estimates of parametric uncertainty. A later document (Rood 2000b) expanded the deterministic analysis to three cover designs. Recent environmental monitoring data (US Ecology 1999) have detected radionuclides in the subsurface. Because of the way the original transport model in Rood (2000a) and Rood (2000b) was constructed, evaluation of radionuclide concentrations in the unsaturated zone was not possible. Additional information regarding waste disposal history, the effects of open trenches on water infiltration, and evolution of ideas regarding the site conceptual model led WDOH to revisit the groundwater assessment performed for the DEIS.

This report documents a reassessment of groundwater concentration estimates including an uncertainty analysis, for radionuclides disposed at the US Ecology low-level radioactive waste disposal facility. This assessment includes an evaluation of three cover designs and several closure dates. The simulations incorporate recent field data taken at the site, along with a conceptual model that includes both historical and future waste disposals. The primary objective of this work was to provide estimates of groundwater concentrations as a function of time for radionuclides disposed at the US Ecology site and to reconcile radionuclide measurements in the unsaturated zone with model estimates. These estimates were intended to error on the side of conservatism. The transport models used in this assessment are relatively simple, but incorporate the major processes that govern the release and transport of radionuclides from the disposal trenches to the aquifer. Estimates of radiation dose from the consumption of drinking water were also made. This work was funded by the WDOH under contract number N08344.

FACILITY DESCRIPTION AND HISTORY

This section provides a brief description of the US Ecology site, its historical operations, and the geology, hydrology, and climate of the Hanford area. It focuses only on the salient features that are pertinent to this assessment. A more detailed description can be found in Kincaid et al. (1998), and US Ecology (1994).

The US Ecology site is located on the Hanford Reservation in Eastern Washington State near the city of Richland. The site is located between the Department of Energy (DOE) 200 Area West and 200 Area East facilities, near the southwest corner of 200 Area East (Figure1). The US Ecology disposal site began operation in 1965 with the opening of Trench Number 1 (Figure 2, Table 1) followed by 19 other trenches. The early trenches were left open as they were filled with soil being placed over the trench as a new trench was excavated to receive waste shipments. A separate trench was set aside to receive chemical waste. Three yet-to-be-dug trenches have been

proposed to receive future waste (Trenches 17, 19, and 20) and three trenches are currently open (12-A, 15, 18).

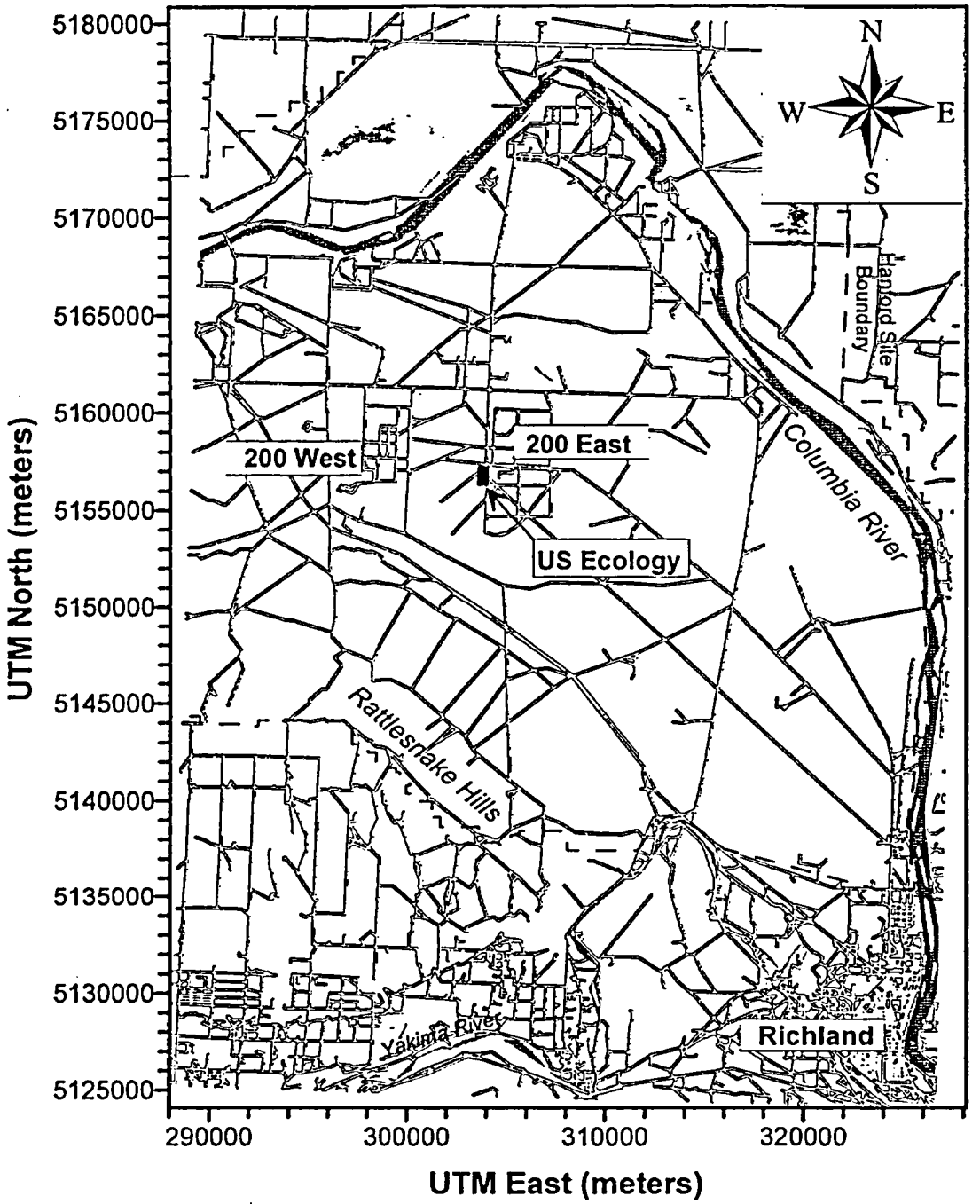

Figure 1. Location of the US Ecology site within the Hanford Reservation in eastern Washington State. The 200 East and 200 West area, the Columbia and Yakima Rivers, and north Richland are also shown. Groundwater flow is generally to the north and east from the Rattlesnake Hills toward the Columbia River. The coordinate system used in the map is the Universal Transverse Mercator (UTM).

In 1999, a comprehensive facility investigation was completed for the US Ecology site (US Ecology 1999). Data obtained from this investigation included concentrations of radionuclides in soil borings taken below Trench 5. These measurements were used to calibrate the unsaturated transport model.

Geology, Hydrology, and Climate

The Hanford Site lies within the Pasco Basin, a structural depression that has accumulated a relatively thick sequence of fluvial, lacustrine, and glacio-fluvial sediments (Kincaid et al. 1998). Underlying the fluvial and lacustrine sediments of the Ringold formation and the glacio-fluvial Hanford formation, are a thick sequence of basalts known as the Columbia River Basalt group. Together, the Hanford and Ringold formation host an unconfined aquifer system. The unconfined system is greater than 61 m in some locations, but its thickness decreases near the flanks of basalt ridges that lie to the west of the site. Groundwater flow is generally from recharge areas in the west toward the Columbia River to the north and east. Transmissivity in the aquifer varies from ~100 m^2 d^{-1} up to 92,900 m^2 d^{-1} (Figure 4.18 in Kincaid et al. 1998). Near the US Ecology site, transmissivity ranged from ~465 to 12,500 m^2 d^{-1}. The unconfined aquifer system is the point of compliance for this assessment.

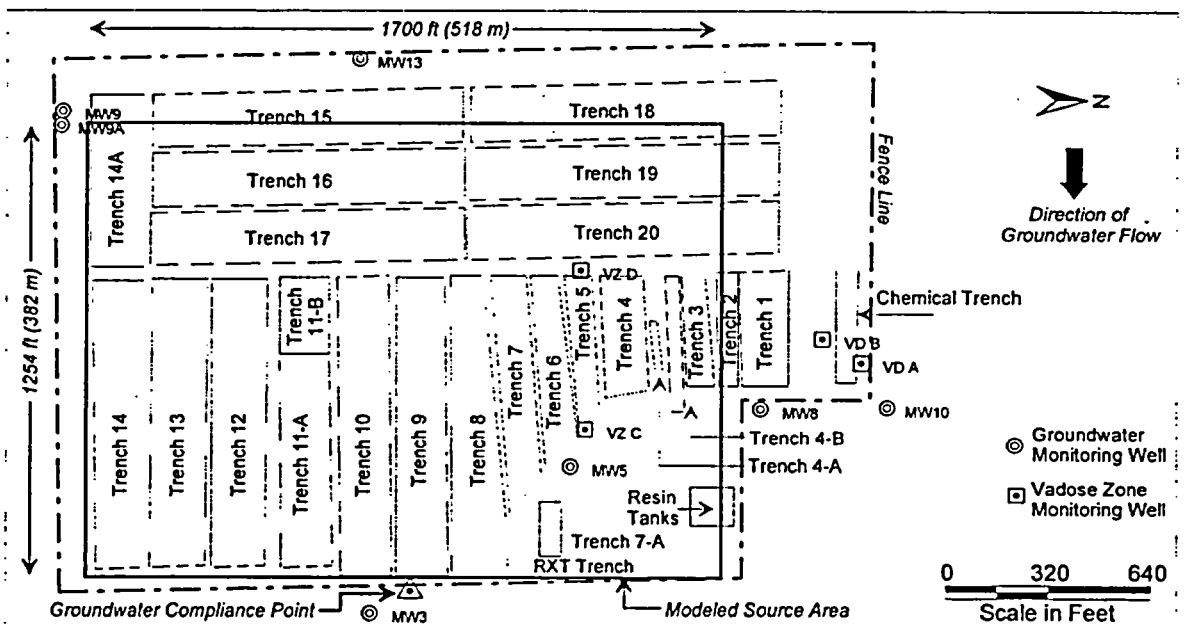

Figure 2. Map of US Ecology facility showing trench locations, property boundaries, monitoring wells points, modeled source area, and the groundwater compliance point (redrawn from Figure 1 in US Ecology 1999).

Soils in the 200 Area of Hanford are predominately course-textured alluvial sands, covered by a variable thick mantel of wind-borne fine sands (Gee at al. 1992). Gravel contents range from 2 to 43% (Kincaid et al. 1998). The soils have potentially high infiltration capacities. The 79-year annual average precipitation at the Hanford site is 16.2 cm yr^{-1} (Gee et al. 1992). Winters are

typically cool and wet while hot and dry conditions persist during summer months. Consequently, most of the water available for recharge comes from winter precipitation when evapotranspiration rates are low. Annual recharge rates range from near zero to about 100-mm yr^{-1} (Gee et al. 1992) and are highly dependent on soil type and vegetative cover. Recent estimates of infiltration for course sediments on a vegetation-free surface are around 7.5 cm yr^{-1} and 0.5 cm yr^{-1} for a vegetated surface (Kincaid et al. 1998).

Table 1. Trench Open and Close Dates for the US Ecology Site (Data provided by WDOH)

Trench Identification	Open Date	Close Date
1	Sep-65	Sep-66
2	Aug-66	Nov-71
3	Dec-71	Mar-75
4	Apr-75	Aug-78
4-A	Apr-82	Jun-82
4-B	Jul-84	Aug-85
5	Apr-78	Sep-79
6	Aug-79	Jun-80
7	Oct-82	Oct-83
7-A	Jun-85	Jul-85
8	May-80	May-81
9	Sep-83	Nov-84
10	May-81	Dec-82
11-A	Oct-84	Jan-86
11-B	Oct-84	Open
12-A	Aug-99	Sep-99
13	Jul-85	Mar-95
14	Feb-87	Open
15	Proposed	Proposed
16	Jan-92	Jun-99
17	Proposed	Proposed
18	Nov-95	Open
19	Proposed	Proposed
20	Proposed	Proposed

RADIONUCLIDE INVENTORIES

Radionuclide inventories were provided by WDOH in two Microsoft Excel® spreadsheets and one Microsoft Word® Document. The primary source term data were obtained from the spreadsheet "sourceterm.xls" and included data on 26 radionuclides plus naturally occurring uranium, naturally occurring thorium, and depleted uranium for disposals from 1965 to 2002. Disposals during the 1965-2002 were segregated into 12 time periods; 1965-1981, 1982-1987, 1988-1993, 1994, 1995, 1996, 1997, 1998, 1999, 2000, 2001, and 2002. These values were supplemented with data from the Microsoft Word® file, "potential additional isotopes for gw modeling.doc". Nineteen radionuclides were listed in this document, but some were already

included in "sourceterm.xls". Additional isotopes were listed by total disposal inventories from 1965 to 2002 and were not segregated into annual disposal amounts. The future projected annual disposals for 21 radionuclides were provided in the spreadsheet "Source Term projections for Art 101302.xls".

Table 2 shows the estimated radionuclide inventory disposed of in the US Ecology site for the time periods, 1965-2002, 1965-2056, and 1965-2215. The values are not decay corrected, but represent the total activity disposed for these time periods. Additionally, values have been rounded to two significant digits, so the sum of the proposed and 1965-2002 inventory may not exactly add up to the values listed in Total column in Table 2. The time periods represent the three closure options considered by WDOH; that is, closure in year 2003, closure in year 2056, and closure in year 2215. Time-variable disposal rates are presented in a later section and were considered in the detailed modeling for those radionuclides that were not removed from consideration through a screening process that is explained in the next section.

Inventory values for the uranium isotopes (U-238, U-235, and U-234) from 1965 to 2002 were later revised by WDOH[1] from the original values provided in the spreadsheet "sourceterm.xls". Originally, uranium was segregated into the three primary isotopes (U-234, U-235, and U-238) plus natural uranium, and depleted uranium. The revised uranium numbers provided in the spreadsheet, "Recommended uranium values for USE.xls" were only segregated by uranium isotope. Estimates of future disposals of U-235 and U-234 were also revised from the original values provided by WDOH in the spreadsheet "sourceterm.xls" because the U-235 activity exceeded that of U-238 and no U-234 values were provided. The projected activity disposal rates for U-235 and U-234 were calculated by multiplying the U-238 proposed activity disposal rate by the ratio of the 1965–2002 activity disposed for U-235 and U-234 respectively to the corresponding U-238 value.

Table 2. Radioactive Inventories for the U.S. Ecology Low-Level Waste Site (Data provided by WDOH)

Radionuclide	Inventory 1965-2002 (mCi)[a]	Additional Isotopes 1965-2002 (mCi)[b]	Proposed (mCi yr^{-1})[c]	Total 1965-2056 (mCi)	Total 1965-2215 (mCi)
Ac-227	6.01E+00			6.01E+00	6.01E+00
Am-241	4.64E+05		5.59E+01	4.67E+05	4.76E+05
Ba-133		6.68E+03		6.68E+03	6.68E+03
Bi-207		1.17E+03		1.17E+03	1.17E+03
C-14	3.97E+06		2.07E+04	5.09E+06	8.37E+06
Cd-113		2.94E+03		2.94E+03	2.94E+03
Cl-36	3.12E+03		2.05E+00	3.23E+03	3.55E+03
Cm-244		2.08E+05		2.08E+05	2.08E+05
Co-60	1.53E+09			1.53E+09	1.53E+09
Cs-134		1.59E+07		1.59E+07	1.59E+07
Cs-137		1.21E+08		1.21E+08	1.21E+08
Eu-152		2.52E+06		2.52E+06	2.52E+06

[1] Revised uranium inventory numbers were provided by Drew Thatcher (WDOH), January 7, 2003 in the spreadsheet, "Recommended uranium values for USE.xls".

Table 2. Radioactive Inventories for the U.S. Ecology Low-Level Waste Site (Data provided by WDOH)

Radionuclide	Inventory 1965-2002 (mCi)[a]	Additional Isotopes 1965-2002 (mCi)[b]	Proposed (mCi yr^{-1})[c]	Total 1965-2056 (mCi)	Total 1965-2215 (mCi)
Eu-154		2.14E+06		2.14E+06	2.14E+06
Eu-155		4.48E+04		4.48E+04	4.48E+04
Fe-55		2.78E+08		2.78E+08	2.78E+08
H-3	7.99E+08		1.12E+06	8.60E+08	1.04E+09
Hf-182		1.56E+03		1.56E+03	1.56E+03
I-129	5.63E+03		6.35E+00	5.98E+03	6.99E+03
K-40	4.76E+03			4.76E+03	4.76E+03
Kr-85		5.89E+07		5.89E+07	5.89E+07
Na-22		3.47E+04		3.47E+04	3.47E+04
Nb-94	7.09E+03		5.95E+01	1.03E+04	1.98E+04
Ni-59	1.17E+06		1.94E+04	2.22E+06	5.30E+06
Ni-59 (activated metal)	3.04E+02			3.04E+02	3.04E+02
Ni-63	1.92E+08		3.22E+06	3.66E+08	8.78E+08
Ni-63 (activated metal)	5.40E+06			5.40E+06	5.40E+06
Pa-231	1.31E+00			1.31E+00	1.31E+00
Pb-210	1.92E+04			1.92E+04	1.92E+04
Pm-147		2.94E+08		2.94E+08	2.94E+08
Pu-238	1.06E+07		1.41E+02	1.06E+07	1.06E+07
Pu-239	4.50E+06		1.54E+02	4.51E+06	4.53E+06
Pu-240	1.95E+06		3.67E-03	1.95E+06	1.95E+06
Pu-241	2.48E+07		9.44E+03	2.53E+07	2.68E+07
Pu-242	2.39E+05		1.73E+00	2.39E+05	2.40E+05
Ra-226	2.33E+05		1.67E+03	3.23E+05	5.89E+05
Sb-125		4.17E+06		4.17E+06	4.17E+06
Sm-151		3.19E+03		3.19E+03	3.19E+03
Sr-90	4.44E+07		9.98E+04	4.98E+07	6.57E+07
Tc-99	5.01E+04		9.27E+01	5.51E+04	6.98E+04
Th-230	1.95E+03			1.95E+03	1.95E+03
Th-232	1.16E+04		1.04E+01	1.22E+04	1.38E+04
Th-natural	1.98E+05			1.98E+05	1.98E+05
Tl-204		6.12E+03		6.12E+03	6.12E+03
U-232		1.34E+03		1.34E+03	1.34E+03
U-234	2.79E+05		1.62E+01	2.79E+05	2.82E+05
U-235	3.05E+04		1.77E+00	3.06E+04	3.09E+04
U-238	1.51E+06		8.74E+01	1.51E+06	1.52E+06

a. From "sourceterm.xls" spreadsheet. Values for U-238, U-235, and U-234 were later revised in the spreadsheet "Recommended uranium values for USE.xls".

b. From the document, "Potential additional isotopes for gw modeling.doc"

c. From the spreadsheet "Source term projections for Art 101302.xls". Value for U-234 and U-235 were modified as discussed in text.

Radionuclide Screening

Screening is defined here as an assessment of the potential for a radionuclide to contribute significantly to the overall dose via the groundwater pathway. The purpose of screening is to remove from consideration those radionuclides that do not have the potential to contribute significantly to the overall dose, and thereby focus resources on those radionuclides that are truly important. Screening calculations should be relatively simple, conservative estimates of the dose-potential of a radionuclide. Conservative is defined here as an upper-bound estimate that is intended to overstate the potential for dose. A radionuclide is termed "screened" if it has been removed from the list of important radionuclides following a screening calculation. A radionuclide is termed "not screened" if it has not been removed from the list of important radionuclides following a screening calculation.

Screening Methods

Screening was performed in two phases. In the first phase, a conservative estimate of the water travel time from the disposal site to the aquifer was compared with the radionuclide half-life. If the half-life was less than or equal to $1/10^{th}$ the conservatively estimated water travel time, then the nuclide was screened or removed from further consideration. The $1/10^{th}$ value of the water travel time was chosen because this would assure that all nuclides that were screened (i.e., had half-lives less than $1/10^{th}$ the water travel time) would have gone through a minimum of 10 half-lives before reaching the aquifer, and therefore only $\exp(-\ln(2) \times 10) = 9.7656 \times 10^{-4}$ of their initial inventory would reach the aquifer.

The second phase of screening used the GWSCREEN code (Rood 1999) with conservative transport parameters to estimate the peak annual groundwater ingestion dose to a persons who's drinking water source is an aquifer well immediately down gradient from the US Ecology facility. The peak dose is compared to a dose limit of 4 mrem yr^{-1} committed effective dose equivalent (CEDE) assuming 2 liters of water are ingested per day for 365 days per year. The 4 mrem yr^{-1} CEDE limit is based on the maximum contaminant limit (MCL) of 4 mrem yr^{-1} committed dose equivalent (CDE) for beta-gamma radionuclides as stated in the Code of Federal Regulation 40 CFR 141. Those radionuclides with doses less than 4 mrem yr^{-1} CEDE were removed from further consideration. Drinking water ingestion doses were calculated using the highest dose conversion factor reported by International Commission on Radiation Protection (ICRP) in their CD version of the ICRP database of dose coefficients (ICRP 1998) which is based on the methodology presented in ICRP-67. (ICRP 1993). However, the MCL is based on the committed dose equivalent using data from National Bureau of Standards Handbook 69 which is derived from methodology developed in ICRP 2 (ICRP 1958). The two dose estimates are not entirely comparable and result in different values for the MCL. However, the use of 4 mrem yr^{-1} CEDE as a screening cutoff is still applicable because annual dose limits for low-level waste performance are also based on the CEDE. Therefore, we have adopted the 4 mrem yr^{-1} CEDE as our screening cutoff. To address possible cumulative impacts from nuclides that have doses less than 4 mrem yr^{-1}, the percent contribution to the total dose was also computed. The total dose was computed by summing the maximum dose regardless of the time of maximum. If a screened radionuclide (i.e., a radionuclide with a screening dose of < 4 mrem yr^{-1}) contributed more than 0.1% to the total dose, then it was removed from the screened list and retained for further consideration.

Phase I Screening

Phase I screening required a conservative estimate of the mean unsaturated water travel time and compared this value to the radionuclide half-lives. The mean unsaturated water travel time is given by

$$T_{unsat} = \frac{x\theta}{I} \quad (1)$$

where

T_{unsat} = mean unsaturated water travel time (yr)
x = depth to the aquifer (m)
θ = moisture content in the unsaturated zone ($m^3\ m^{-3}$)
I = infiltration rate ($m\ yr^{-1}$)

A conservative estimate of the site-specific infiltration rate at the US Ecology site was chosen to be 10 cm yr^{-1} based on the observations and measurements in Gee et al. (1992). The depth to the aquifer of 82.3 m was taken from the original DEIS groundwater assessment (Rood 2000a). The moisture content of 0.0606 $m^3\ m^{-3}$ was calculated for 10-cm yr^{-1} infiltration using the moisture characteristic curve presented in Rood (2000a). Using these values in Equation (1) yields a mean unsaturated water travel time of 49.87 yr. One-tenth this value (4.987 yr) was compared to the radionuclide half-life ($T_{1/2}$). If $T_{1/2} \leq T_{unsat}/10$ then the radionuclide was removed from further consideration. That is, if the half-life was less than one-tenth the conservative estimate of the unsaturated *water* travel time, then the radionuclide was eliminated from further consideration or screened. Results of this screening are presented in Table 3.

The radionuclides Cd-113, Hf-182, and Kr-85 were eliminated from consideration because ingestion dose conversion factors were not available. Lack of an ingestion dose conversion factor for a radionuclide indicates ingestion doses are inconsequential or improbable.

Table 3. Phase I Screening Results for the U.S. Ecology Site

Radionuclide	Half-Life (yr)	$T_{1/2} \leq 4.987$ yr?	Radionuclide	Half-Life (yr)	$T_{1/2} \leq 4.987$ yr?
Ac-227	21.773	No	Ni-63AM	100.1	No
Am-241	432.7	No	Pa-231	3.28E+04	No
Ba-133	10.52	No	Pb-210	22.3	No
Bi-207	32.2	No	Pm-147	2.6234	Yes
C-14	5730	No	Pu-238	87.4	No
Cd-113	9.30E+15	No	PU-239	24119	No
Cl-36	3.01E+05	No	Pu-240	6563	No
Cm-244	18.1	No	Pu-241	14.35	No
Co-60	5.2714	No	Pu-242	3.73E+05	No
Cs-134	2.062	Yes	Ra-226	1600	No
Cs-137	30.1	No	Sb-125	2.73	Yes
Eu-152	13.542	No	Sm-151	90	No
Eu-154	8.592	No	Sr-90	29.1	No
Eu-155	4.68	Yes	Tc-99	2.11E+05	No
Fe-55	2.73	Yes	Th-230	7.54E+04	No
H-3	12.33	No	Th-232	1.41E+10	No
Hf-182	9.00E+06	No	Th-nat	1.41E+10	No
I-129	1.57E+07	No	Tl-204	3.78	Yes
K-40	1.28E+09	No	U-232	68.9	No
Kr-85	10.756	No	U-234	2.45E+05	No
Na-22	2.6088	Yes	U-235	7.04E+08	No
Nb-94	2.03E+04	No	U-238	4.47E+09	No
Ni-59	7.50E+04	No	U-dep	4.47E+09	No
Ni-59AM	7.50E+04	No	U-DEP	4.47E+09	No
Ni-63	100.1	No	U-nat	4.47E+09	No

Phase II Screening

The radionuclides that were not screened in Phase I were evaluated in Phase II. As was done in Phase I screening, a conservative infiltration rate of 10-cm yr^{-1} was assumed. Partition coefficient values (K_d) were taken from Kincaid et al. (1998). Most other parameters (Table 4) were taken from Rood (2000a). The partitioning coefficient values that were used represented the most conservative values for source areas (i.e. areas where radionuclides were disposed or discharged into the soil) and unsaturated/aquifer materials as reported in Kincaid et al. (1998). Source-area partition coefficients were assumed to represent a highly mobile environment and in many cases, were near zero. Partition coefficients were not available in Kincaid et al. (1998) for all radionuclides considered. For the radionuclides not available in Kincaid et al. (1998), a K_d value of 0 mL g^{-1} was assumed for the source and the lowest K_d value reported in Sheppard and Thibault (1990) was used for unsaturated/aquifer materials. Partition coefficient values were then corrected for the percent gravel composition in the unsaturated zone and aquifer (see Equation 12

and Table 10). The simulation did not consider waste emplacement rates over time. Instead, the entire inventory was assumed to be placed in the trenches at the start of the simulation. This assumption provides the most conservative estimate of the maximum mass flux from the source area to the aquifer.

Table 4. Parameter used in the Phase II Screening Analysis using GWSCREEN

Parameter name (units)	Value
Source length (m)	518
Source width (m)	382
Percolation (m yr^{-1})	0.1
Source thickness (m)	10.6
Bulk density of source (g cm^{-3})	1.26
Moisture content in source zone (m^{-3} m^{-3})[a]	0.0606
Unsaturated zone thickness (m)	82.3
Bulk density of unsaturated zone (g cm^{-3})	1.6
Unsaturated zone dispersivity (m)	0
Percent gravel in unsaturated zone and aquifer	41.7%
Moisture content in unsaturated zone (m^{-3} m^{-3})[a]	0.0606
Longitudinal dispersivity in aquifer (m)	27.5
Transverse dispersivity in aquifer (m)	5.0
Bulk density of aquifer (g cm^{-3})	1.6
Aquifer porosity (m^{-3} m^{-3})	0.1
Darcy velocity in aquifer (m y^{-1})	32.9
Receptor distance (m)[b]	275

a. Calculated using van Genuchten fitting parameters in Rood 2000a: $\alpha = 7.51$ m^{-1}, $n = 2.298$, $K_{sat} = 1710$ m y^{-1}, $\theta_{sat} = 0.2724$, $\theta_{residual} = 0.0321$

b. Measured from the center of the source. Transverse distance = 0 m.

Ingrowth of radioactive progeny was also considered for actinides. For some actinides that are relatively immobile, have short-half lives relative to their transit time in the unsaturated zone, and have long-lived mobile progeny, transport of the progeny was modeled instead of that of the parent. Nuclides that fall into this category include Am-241→Np-237, Cm-244→Pu-240, Pu-238→U-234, and Pu-241→Am-241→Np-237. In these cases, the parent activity was conservatively converted into the equivalent mobile progeny activity by multiplying the parent activity by the ratio of the progeny/parent half-lives.

For actinides with relatively short-lived progeny (≤ 1 year), parent and progeny were assumed to be in secular equilibrium and the dose conversion factors were summed as shown in Table 5.

Table 5. ICRP Dose Conversion Factors for Actinide Decay Chains[a]

Parent	Progeny	Progeny Included	Subtotals (rem Ci^{-1})	Total (rem Ci^{-1})	Total (mrem pCi^{-1})
Pu-242			8.88E+05	8.88E+05	8.88E-04
	U-238		1.67E+05		
		Th-234	1.26E+04		
		Pa-234	1.89E+03		

Table 5. ICRP Dose Conversion Factors for Actinide Decay Chains[a]

Parent	Progeny	Progeny Included	Subtotals (rem Ci^{-1})	Total (rem Ci^{-1})	Total (mrem pCi^{-1})
		Total	1.81E+05	1.81E+05	1.81E-04
	U-234		1.81E+05	1.81E+05	1.81E-04
	Th-230		7.77E+05	7.77E+05	7.77E-04
	Ra-226		1.04E+06		
		Pb-214	5.18E+02		
		Bi-214	4.07E+02		
		Total	1.04E+06	1.04E+06	1.04E-03
	Pb-210		2.55E+06		
		Bi-210	4.81E+03		
		Po-210	4.44E+06		
		Total	7.00E+06	7.00E+06	7.00E-03
Pu-241			1.78E+04	1.78E+04	1.78E-05
	Am-241		7.40E+05	7.40E+05	7.40E-04
	Np-237		4.07E+05		
		Pa-233	3.22E+03		
		Total	4.10E+05	4.10E+05	4.10E-04
	U-233		1.89E+05	1.89E+05	1.89E-04
	Th-229		1.81E+06		
		Ra-225	3.66E+05		
		Ac-225	8.88E+04		
		Bi-213	7.40E+02		
		Total	2.27E+06	2.27E+06	2.27E-03
Pu-240			9.25E+05	9.25E+05	9.25E-04
	U-236		1.74E+05	1.74E+05	1.74E-04
	Th-232		8.51E+05	8.51E+05	8.51E-04
	Ra-228		2.55E+06		
		Ac-228	1.59E+03		
		Total	2.55E+06	2.55E+06	2.55E-03
	Th-228		2.66E+05		
		Ra-224	2.41E+05		
		Pb-212	2.22E+04		
		Bi-212	9.62E+02		
		Total	5.30E+05	5.30E+05	5.30E-04
Pu-239			9.25E+05	9.25E+05	9.25E-04
	U-235		1.74E+05		
		Th-231	1.26E+03		
		Total	1.75E+05	1.75E+05	1.75E-04
	Pa-231		2.63E+06	2.63E+06	2.63E-03
	Ac-227		4.07E+06		
		Th-227	3.26E+04		
		Fr-223	8.88E+03		
		Ra-223	3.70E+05		
		Pb-211	6.66E+02		
		Total	4.48E+06	4.48E+06	4.48E-03

[a] From the ICRP Database of Dose Coefficients, version 1.0 (ICRP 1998). Original units were Sv Bq^{-1} and were converted to rem Ci^{-1} and mrem pCi^{-1}.

For Ni-59 and Ni-63, WDOH segregated activated metal waste forms from the remainder of the inventory. The activated metal inventory was about two orders of magnitude lower than that

of the other inventory for these nuclides presented in Table 1, and therefore were considered insignificant in terms of the overall inventory for screening purposes.

Phase II screening results are presented in Table 6. Those radionuclides that are retained for the final analysis are shaded. Fifteen radionuclides (compared to 5 in the original analysis) were not screened and retained for further evaluation: C-14, Cl-36, H-3, I-129, Pu-238,-239,-240,-242, Ra-226, Tc-99, Th-230, Th-232, U-234, U-235, and U-238. Strontium-90 and Ni-63 were screened from radionuclide inventory but were retained because these radionuclides had detectable concentrations in the unsaturated zone below trench 5.

Table 6. Results of Phase II Screening
(Shaded radionuclides were not screened and retained for further evaluation)

Radionuclide/ Progeny[a]	Number of progeny	Half-Life (yr)[b]	DCF (mrem pCi^{-1})[c]	Source K_d (mL g^{-1})[d]	Unsaturated/ aquifer K_d (mL g^{-1})[e]	Total Dose (mrem yr^{-1})[f]	Is Dose<4 mrem yr^{-1} ?	% of total
Ac-227	0	2.18E+01	4.48E-03	0	100	0.00E+00	Yes	0.000%
Am-241[Np]	2	2.14E+06	4.10E-04	0.1	10	1.19E+00	Yes	0.002%
U-233	na	1.59E+05	1.89E-04	0	0.6			
Th-229	na	7.43E+03	2.27E-03	0	40			
Ba-133	0	1.05E+01	5.55E-06	0	60	0.00E+00	Yes	0.000%
Bi-207	0	3.22E+01	4.81E-06	0	100	0.00E+00	Yes	0.000%
C-14	0	5.73E+03	2.15E-06	0	0	8.44E+03	No	12.937%
Cl-36	0	3.01E+05	3.44E-06	0	0	5.77E+00	No	0.009%
Cm-244(Pu)	4	6.56E+03	9.25E-04	0.1	80	1.12E-02	Yes	0.000%
U-236	0	2.34E+07	1.74E-04	0	0.6			
Th-232	0	1.41E+10	8.51E-04	0	40			
Ra-228	0	5.75E+00	2.55E-03	0	8			
Th-228	0	1.91E+00	5.30E-04	0	40			
Co-60	0	5.27E+00	1.26E-05	0	1200	0.00E+00	Yes	0.000%
Cs-137	0	3.01E+01	4.81E-05	5	540	0.00E+00	Yes	0.000%
Eu-152	0	1.35E+01	5.18E-06	0	100	0.00E+00	Yes	0.000%
Eu-154	0	8.59E+00	7.40E-06	0	100	0.00E+00	Yes	0.000%
H-3	0	1.23E+01	1.55E-07	0	0	3.97E+03	No	6.093%
I-129	0	1.57E+07	4.07E-04	0	0.3	9.09E+02	No	1.393%
K-40	0	1.28E+09	2.29E-05	0	15	1.84E+00	Yes	0.003%
Nb-94	0	2.03E+04	6.29E-06	2	50	1.34E-01	Yes	0.000%
Ni-59	0	7.50E+04	2.33E-07	2	50	3.55E+00	Yes	0.005%
Ni-63[g]	0	1.00E+02	5.55E-07	2	50	0.00E+00	Yes	0.000%
Pa-231	1	3.28E+04	2.63E-03	0.1	10	8.37E-02	Yes	0.000%
Ac-227	na	2.18E+01	4.48E-03	0	100			
Pb-210	0	2.23E+01	2.55E-03	0	2000	0.00E+00	Yes	0.000%
Pu-238[U]	3	2.45E+05	1.81E-04	0.1	0.6	8.75E+01	No	0.134%
Th-230	na	7.54E+04	7.77E-04	0	40			
Ra-226	na	1.60E+03	1.04E-03	0	8			
Pb-210	na	2.23E+01	7.00E-03	0	2000			
Pu-239	3	2.41E+04	9.25E-04	0.1	80	2.22E+03	No	3.402%

Table 6. Results of Phase II Screening
(Shaded radionuclides were not screened and retained for further evaluation)

Radionuclide/ Progeny[a]	Number of progeny	Half-Life (yr)[b]	DCF (mrem pCi^{-1})[c]	Source K_d (mL g^{-1})[d]	Unsaturated/ aquifer K_d (mL g^{-1})[e]	Total Dose (mrem yr^{-1})[f]	Is Dose<4 mrem yr^{-1} ?	% of total
U-235	na	7.04E+08	1.75E-04	0	0.6			
Pa-231	na	3.28E+04	2.63E-03	0	10			
Ac-227	na	2.18E+01	4.48E-03	0	100			
Pu-240	4	6.56E+03	9.25E-04	0.1	80	4.19E+01	No	0.064%
U-236	na	2.34E+07	1.74E-04	0	0.6			
Th-232	na	1.41E+10	8.51E-04	0	40			
Ra-228	na	5.75E+00	2.55E-03	0	8			
Th-228	na	1.91E+00	5.30E-04	0	40			
Pu-241[Np]	2	2.14E+06	4.10E-04	0.1	10	2.23E+00	Yes	0.003%
U-233	na	1.59E+05	1.89E-04	0	0.6			
Th-229	na	7.43E+03	2.27E-03	0	40			
Pu-242	5	3.73E+05	8.88E-04	0.1	80	6.04E+02	No	0.926%
U238	na	4.47E+09	1.81E-04	0	0.6			
U-234	na	2.45E+05	1.81E-04	0	0.6			
Th-230	na	7.54E+04	7.77E-04	0	40			
Ra-226	na	1.60E+03	1.04E-03	0	8			
Pb-210	na	2.23E+01	7.00E-03	0	2000			
Ra-226	1	1.60E+03	1.04E-03	0.1	8	1.21E+03	No	1.856%
Pb-210	na	2.23E+01	7.00E-03	0	2000			
Sm-151	0	9.00E+01	3.63E-07	0	245	0.00E+00	Yes	0.000%
Sr-90[g]	0	2.91E+01	1.04E-04	0.1	8	1.20E-60	Yes	0.000%
Tc-99	0	2.11E+05	2.37E-06	0	0	7.81E+01	No	0.120%
Th-230	2	7.54E+04	7.77E-04	1	40	5.18E+01	No	0.079%
Ra-226	na	1.60E+03	1.04E-03	0	8			
Pb-210	na	2.23E+01	7.00E-03	0	2000			
Th-232	2	1.41E+10	8.51E-04	1	40	1.09E+03	No	1.668%
Ra-228	na	5.75E+00	2.55E-03	0	8			
Th-228	na	1.91E+00	5.30E-04	0	40			
U-232	1	6.89E+01	1.22E-03	0.1	0.6	1.04E+00	Yes	0.002%
Th-228	na	1.91E+00	5.30E-04	0	40			
U-234	3	2.45E+05	1.81E-04	0.1	0.6	6.23E+02	No	0.956%
Th-230	na	7.54E+04	7.77E-04	0	40			
Ra-226	na	1.60E+03	1.04E-03	0	8			
Pb-210	na	2.23E+01	7.00E-03	0	2000			
U-235	2	7.04E+08	1.75E-04	0.1	0.6	4.82E+03	No	7.387%
Pa-231	na	3.28E+04	2.63E-03	0	10			
Ac-227	na	2.18E+01	4.48E-03	0	100			
U-238	4	4.47E+09	1.81E-04	0.1	0.6	4.11E+04	No	62.960%
U-234	na	2.45E+05	1.81E-04	0	0.6			
Th-230	na	7.54E+04	7.77E-04	0	40			

Table 6. Results of Phase II Screening
(Shaded radionuclides were not screened and retained for further evaluation)

Radionuclide/ Progeny[a]	Number of progeny	Half-Life (yr)[b]	DCF (mrem pCi^{-1})[c]	Source K_d (mL g^{-1})[d]	Unsaturated/ aquifer K_d (mL g^{-1})[e]	Total Dose (mrem yr^{-1})[f]	Is Dose<4 mrem yr^{-1} ?	% of total
Ra-226	na	1.60E+03	1.04E-03	0	8			
Pb-210	na	2.23E+01	7.00E-03	0	2000			
Total						7.24E+04		

a. Radioactive progeny that are included in the dose estimate are right justified. Radionuclides followed by another radionuclide in brackets (e.g., Am-241[Np-237]) indicates that the radionuclide in brackets was the radionuclide modeled.
b. From Tuli (1990).
c. From ICRP (1998). Contributions from radioactive progeny in secular equilibrium are included in the dose estimate.
d. Partition coefficients for the following elements were obtained from Kincaid et al., 1998: H, Cl, Tc, Ac, Am, Cm, Eu, C, Co, Cs, I, Ni, Sn, Nb, Np, Pa, Pb, Pu, Ra, Sr, Ru, Se, Th, Zr, and U. Source K_d values represent the conservative estimate for Source Term Category A in Table E.5 of Kincaid et al. (1998). Elements not represented in Kincaid et al. (1998) were assumed to be zero. Radioactive progeny are assumed to travel with their parent, therefore no source K_d values are presented for radioactive progeny. Partition coefficient values shown are not corrected for the percent gravel component.
e. Partition coefficient values represent the conservative estimate for Source Term Category F in Table E.10. Elements not represented in Kincaid et al. (1998) were from Sheppard and Thibault (1990). Partition coefficient values shown are not corrected for the percent gravel component.
f. The total dose includes all contributions from progeny. The doses from progeny are not shown. These doses should not be interpreted as a realistic estimate of radiological impacts from the site.
g. These radionuclides had screening doses less than 4 mrem yr^{-1} but were retained because they were detected in the unsaturated zone below Trench 5.

CONCEPTUAL MODEL OF FATE AND TRANSPORT

In this section the conceptual model for radionuclide fate and transport is presented for the three primary elements of the transport model: the source term, the unsaturated zone, and the aquifer (Figure 3). The source term represents the release of radionuclides from the waste and transport through the bottom of the trenches and is geometrically represented by the total volume of trenches within the facility. Individual trenches are not modeled; rather, all the trenches are modeled as a single composite trench that represents the entire disposal facility. The area of the composite trench is illustrated in Figure 2 and labeled "modeled source area". The unsaturated zone represents the area from the bottom of the trenches to the top of the aquifer where the rock matrix is partially saturated and water flow is vertical and downward. The unsaturated zone is composed multiple layers, each having their own unique properties and water fluxes. The aquifer represents a fully saturated media where water flow is essentially horizontal. The three primary elements are linked by radionuclide fluxes across their boundaries. For example, the source term and unsaturated zone are linked by the radionuclide flux from the bottom of the trenches to the top of the unsaturated zone.

Infiltrating water is the primary mechanism of radionuclide transport. Vapor transport is only important for tritium and radon, however tritium is conservatively assumed to move only by aqueous phase transport. Gas-phase radon transport is not considered and radon progeny are assumed to travel with radium. Radionuclides are present in two phases; a sorbed solid phase and a dissolved aqueous phase. Partitioning between the sorbed and aqueous phases is described by the equilibrium partitioning coefficient or K_d. As infiltrating water comes in contact with the waste, radionuclides partition into the aqueous phase according to the K_d and are transported with

the water. Radionuclide pore water concentrations are not allowed to exceed their element-specific solubility limit. Radioactive progeny that form during transport are also accounted for and partition according to their element-specific K_d.

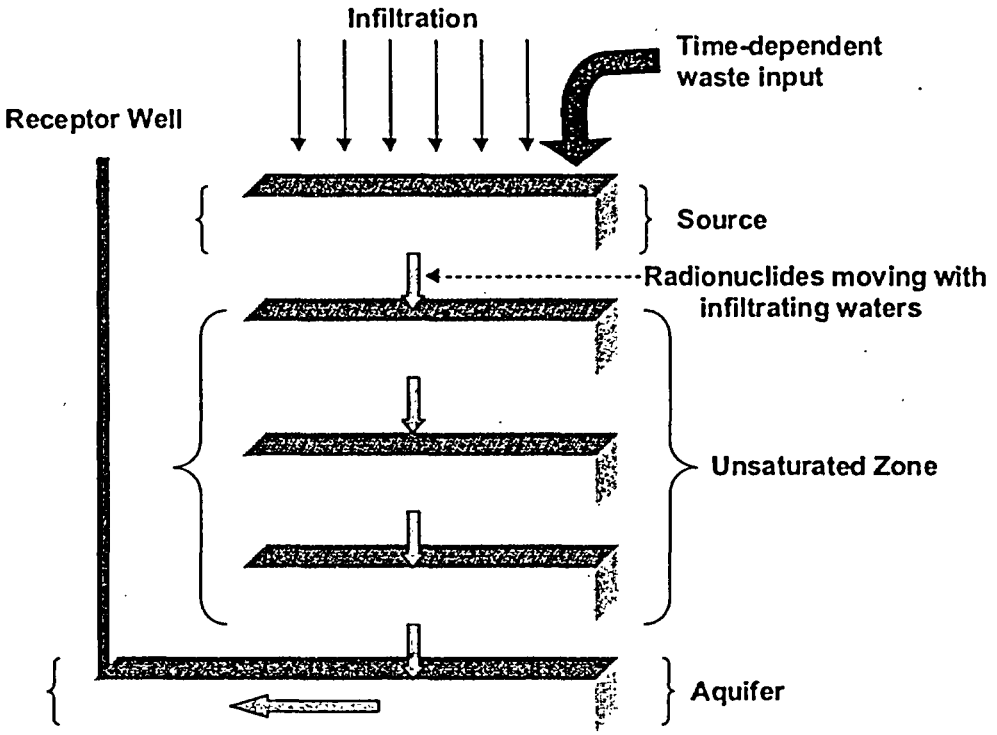

Figure 3. Overall conceptual model for US Ecology LLRW facility showing the three primary elements; source, unsaturated zone, and aquifer. The source is modeled as a separate unit. The unsaturated zone is composed of multiple layers, each having unique properties and water flux.

Source Term and Unsaturated Zone Conceptual Model

The source term conceptual model for the site was segregated into three time periods; pre-cover, cover, and post cover. The pre-cover period begins in 1965 (Figure 4) when the facility started operation and ends when the cover is installed. In all cases, a cover is assumed to be installed in the year 2005. Future operations of the site are assumed to limit infiltration through the active trenches to no more than the designed infiltration rate of the cover. Eventually, water fluxes through the trenches extend vertically down to the aquifer. The temporal histories of waste disposals are accounted for in the model. That is, waste is disposed over time as represented by the disposal history provided by WDOH. While a trench is actively receiving waste, infiltration is enhanced. After the trench is closed (ceases to receive waste), the trench is backfilled with soil. Infiltration through the trench after closure is lower than active disposal, but higher than natural infiltration.

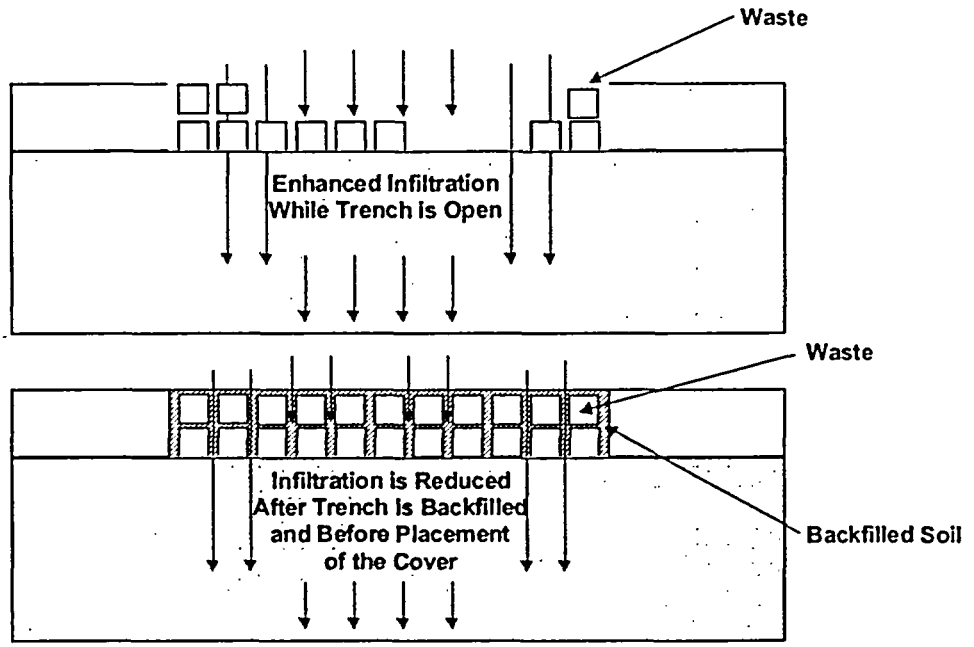

Figure 4. Source term conceptual model of the US Ecology LLRW facility from 1965 to the time of cover placement (2005). Enhanced water infiltration is assumed to occur in the open trench and infiltration slightly enhanced over background occurs after the trench is closed (backfilled with soil) and before placement of the cover.

The cover time period represents the time when the cover is intact and performs according to its design specifications (Figure 5). The cover restricts infiltration through the waste. Some of the precipitation that falls on the cover runs off the sides and infiltrates around the edge of the cover, although this amount is assumed to be minimal. After placement of the cover, soils underneath the facility dry over time. The drying front as it is referred to here advances over time until reaching the aquifer. Once the drying front reaches the aquifer, water fluxes throughout the unsaturated zone are equivalent to the net water infiltration rate through the cover.

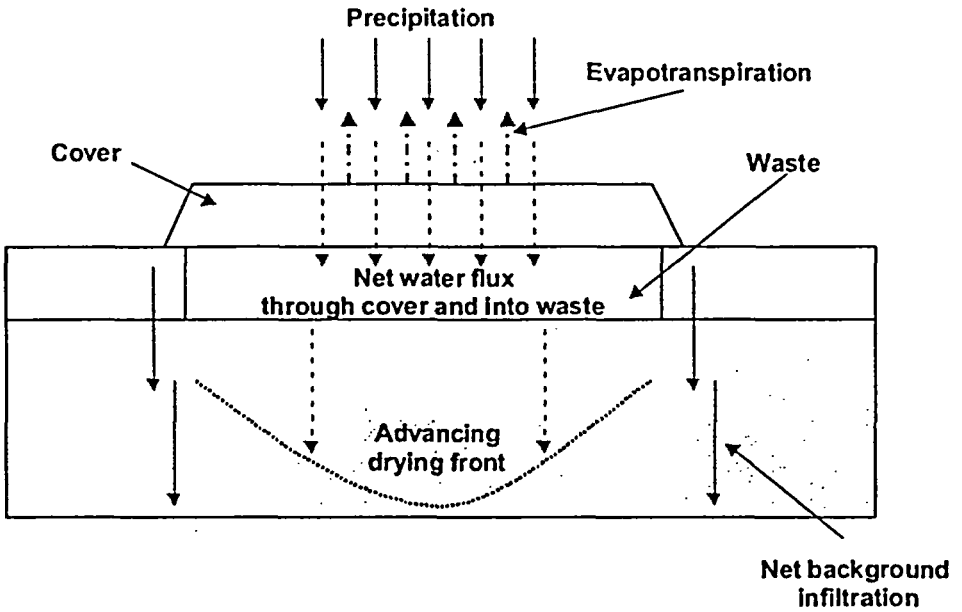

Figure 5. Source term conceptual model of the US Ecology LLRW facility from the time of cover placement (2005) to the time of cover failure. The amount of water that passes through the cover and into the waste is specified by the design of the cover.

The post cover period represents the time when the cover fails and infiltration through the waste returns to natural recharge rates over time (Figure 6). The wetting front advances over time until it reaches the aquifer. Water flux through the cover, waste, and unsaturated zone are assumed to be constant for all future times after the wetting front reaches the aquifer.

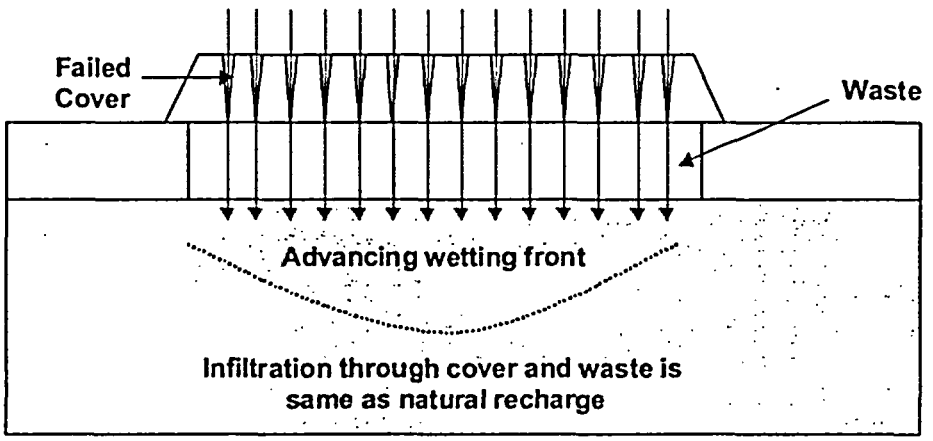

Figure 6. Source term conceptual model of the US Ecology LLRW facility for times after cover failure. Net infiltration through the cover is assumed to be the same as natural recharge.

Waste packaging is assumed to be ineffective in controlling water from coming in contact with the waste. Partitioning between radionuclides in the waste form and infiltrating water, and

between radionuclides in back-filled soil and infiltrating water, is treated as a single process characterized by a single partitioning coefficient.

For modeling purposes, the entire site is represented by a single composite trench as shown in Figure 1. The surface area (identified as "Modeled Source Area" in Figure 1) of the composite trench represents the total surface area of all individual trenches. Infiltration through an open trench is assumed to be greater than infiltration through a closed trench. Therefore, infiltration through the composite trench represents an area-weighted infiltration that is based on the number of open trenches at a given time.

Radionuclides leaving the bottom of the trenches enter the unsaturated zone. The net water flow in the unsaturated zone is assumed to be vertical and downward. Where sufficient data exists, the characteristics of specific lithologic units are accounted for in this model. Radionuclides and radioactive progeny partition between the rock matrix and the infiltrating water according to their element-specific partitioning coefficient. Partitioning was assumed to only occur on the portion of the rock matrix composed of fine material and not on the coarse gravelly components (Kincaid et al. 1998).

Aquifer Conceptual Model

Radionuclides enter the aquifer across an area defined by the footprint of the source as illustrated in Figure 1. The aquifer is assumed to be a homogeneous porous media of infinite lateral extent and finite thickness. Aquifer flow is assumed to be constant and unidirectional, with no appreciable sources or sinks within the footprint of the facility. A drinking water well is assumed to be drilled on the downgradient edge of the facility at the property line. The well is assumed to have a screened interval beginning at the surface of the aquifer and extending 15-m below its surface, the length of a typical well screen. Pumping from the well is assumed to be minimal and have little impact on the overall flow in the aquifer. Ingrowth of radioactive progeny is not considered in the aquifer because transport times from the radionuclide source in the aquifer to the receptor well are relatively short. This simplifying assumption is considered suitable for this analysis where the receptor well is relatively close to the source. However, this assumption is not considered suitable for receptor distances a substantial distance from the source.

IMPLEMENTATION OF THE CONCEPTUAL MODEL

In the first iteration of this assessment (Rood 2000a), the GWSCREEN Version 2.5 code (Rood 1999) was used to implement a simplified version of the conceptual model presented earlier. Although GWSCREEN was suitable for the earlier iteration of this assessment, it lacks the processes necessary to implement the source term and unsaturated conceptual models as outlined in the previous section. Therefore, it was necessary to investigate alternative models for use in this assessment.

Preliminary Modeling with DUST

The Disposal Unit Source Term Model (DUST) code (Sullivan 1996) provided a viable alternative to GWSCREEN for computing the source term and unsaturated transport. DUST is a waste form release model coupled with a one-dimensional finite difference approximation to the advection dispersion equation used to compute transport in the unsaturated zone. DUST allows for time-variable waste disposal rates, container failure rates, and time-variable water fluxes.

Preliminary simulations were performed with DUST for nuclides remaining after Phase II screening. Initial model simulations were satisfactory; however, closer inspection revealed inconsistent mass balance errors ranging from <2% to up to 30% when time-varying waste input rates were considered. Changes in time stepping and finite difference node spacing from their initial values appeared to have little impact on the overall results. However, mass balance errors for initial concentration problems with no time-variable waste input rates were insignificant. Monte Carlo uncertainty analysis was part of this project and it was uncertain whether correct results would be obtained for all model realizations that used time-variable waste input rates. Additionally, implementation of the conceptual model required two iterations of DUST for each radionuclide simulated, further adding to the overall complexity of the simulation. For these reasons, a new model was developed that could implement the conceptual model outlined earlier. For situations where the accuracy of DUST could be assured, it was used as a check on the new model. The GWSCREEN model was retained for radionuclide transport in the aquifer and dose calculation.

Description of the FOLAT Model

The name FOLAT (First Order Leach and Transport) was given to the new model, although non-first order processes may also be included in the model. The FOLAT model treats the source and unsaturated zones as a series of compartments where interchange between the compartments is described by advection-driven first-order or solubility-limited processes. The FOLAT model is conceptually similar to the SESOIL model (Scott and Hetrick 1994) originally developed at Oak Ridge National Laboratory. Details of the FOLAT model are described in a separate document (Rood 2003) and are summarized below.

The conceptual model for FOLAT is relatively simple. The subsurface environment is envisioned to be composed of a series of "compartments". Within each compartment, radionuclides enter, mix, sorb, decay, and are eventually removed by the downward movement of water. Each compartment may have its own unique qualities that include horizontal and vertical

dimensions, bulk density, porosity, hydraulic conductivity, net water flux through the compartment, and sorptive properties. Water flux through each compartment may change as a function of time. As the water flux changes, so too does the moisture content of the compartment. Radionuclides sorb on to the solid matrix as described by the equilibrium partitioning coefficient or K_d. Sorption retards the overall downward movement of radionuclides. The rate of transport of radioactive decay products or progeny that form during vertical transport of a parent radionuclide are governed by the sorptive properties of the progeny, and not those of the parent.

Radionuclides may be present in each of the compartments at the start of the simulation, or alternatively, the parent member of the decay chain may be placed over time in the uppermost compartment. Concentrations of radionuclides in pore water are not allowed to exceed their solubility limit. Unit gradient conditions are assumed to apply to each compartment.

Ordinary differential equations describe the mass balance of radionuclides in each of the compartments. Radionuclide concentrations in pore water and the radionuclide fluxes from each compartment are determined from the radionuclide inventory within each compartment. The uppermost or first compartment for the first (parent) member of the decay chain is described by

$$\frac{dQ_{1,1}}{dt} = R(t) - F_{1,1}(t) - \lambda_1 Q_{1,1} \tag{2}$$

where

$Q_{1,1}$ = the number of atoms in compartment 1 for decay chain member 1 (atoms)
$R(t)$ = the input rate of decay chain member 1 into compartment 1 (atoms time^{-1})
$F_{1,1}(t)$ = the removal rate (flux) of decay chain member 1 from compartment 1 to compartment 2 (atoms time^{-1})
λ_1 = the decay rate constant for decay chain member 1 (time^{-1}).

For simplicity and clarity, all equations are written in terms of the number of atoms of each decay chain member. The mass balance equation for the remaining compartments is given by

$$\frac{dQ_{i,j}}{dt} = F_{i-1,j}(t) - \lambda_j Q_{i,j} + \lambda_{j-1} Q_{i,j-1} - F_{i,j}(t) \tag{3}$$

where i is the index for the compartment and j is the index for the decay chain member and $i \neq 1$ and $j \neq 1$. Other terms are defined previously. Equation 3 assumes the branching ratio between the parent and radioactive progeny is 1.0 (i.e. 100% of the parent decays to the progeny). When the radionuclide concentration in pore water is less than the solubility limit, the flux term in Equations 2 and 3 is given by

$$F_{i,j}(t) = \left(\kappa_{i,j}(t) + \eta_{i,j}\right) Q_{i,j} \tag{4}$$

where

$\kappa_{i,j}(t)$ = the leach rate constant for compartment i and decay chain member j (time^{-1})
$F_{i,j}(t)$ = the flux of decay chain member j from compartment i into compartment i+1 (atoms time^{-1})

$\eta_{i,j}$ = a fixed removal rate constant for compartment i and decay chain member j (time^{-1}).

In general, only the leach rate constant is used to remove radionuclides from a compartment. However, the user may which to bypass this calculation and calculate a removal rate constant outside the code. We have included $\eta_{i,j}$ for this situation. When the pore water concentration exceeds the solubility limit, then the flux term in Equations 2 and 3 is given by

$$F_{i,j}(t) = S_j\, q_i(t)\, L_i\, W_i \tag{5}$$

where

S_j = the solubility limit of decay chain member j (atoms m^{-3})
$q_i(t)$ = water flux through compartment i as a function of time (m time^{-1})
L_i = length of compartment i (m)
W_i = width of compartment i (m).

In Equations 4 and 5, $i \leq n$, where n is the number of compartments in the simulation. Likewise, $j \leq m$, where m is the number of decay chain members including the parent. The pore water concentration in compartment i for decay chain member j ($C_{i,j}$) is given by

$$C_{i,j}(t) = \frac{Q_{i,j}(t)}{\theta_i(t)\, L_i\, W_i\, T_i \left(1 + \dfrac{Kd_{i,j}\,\rho_i}{\theta_i(t)}\right)} \tag{6}$$

where

$\theta_i(t)$ = volumetric moisture content in compartment i as a function of time (m^3 m^{-3})
$Kd_{i,j}$ = equilibrium partition coefficient for compartment i and decay chain member j (mL g^{-1})
ρ_i = bulk density of compartment i (g mL^{-1})
T_i = thickness of compartment i (m)
L_i = length of compartment i (m)
W_i = width of compartment i (m).

The term, $1 + Kd_{i,j}\,\rho_i/\theta(t)_i$ is the retardation coefficient and is 1.0 for a K_d of zero. The leach rate constant is given by Baes and Sharp (1983) as

$$\kappa_{i,j}(t) = \frac{q_i(t)}{\theta_i(t)\, T_i \left(1 + \dfrac{Kd_{i,j}\,\rho_i}{\theta_i(t)}\right)} \tag{7}$$

and the decay rate constant is given by

$$\lambda_j = \frac{\ln(2)}{T1/2_j} \tag{8}$$

where $T1/2_j$ = half-life of decay chain member j.

Transport of radionuclides in the aquifer was computed using the GWSCREEN code. GWSCREEN takes as input, the time-variable radionuclide fluxes from FOLAT at the top of the aquifer and transports them down gradient from the source using a 2 or 3-dimentional semi-analytical solution to the advection-dispersion equation. The 2-dimensional solution was used in this assessment. GWSCREEN assumes radioactive progeny travel at the same rate as their parent. Because transport times from source to receptor are relatively short, generation of progeny in the aquifer was ignored. Concentrations in the aquifer were vertically averaged from the top of the aquifer to the length of a typical well screen.

ENGINEERED COVERS, CLOSURE SCENARIOS AND COMPLIANCE TIME

Five closure and cover design options were simulated in this assessment (Table 7). Water fluxes through the three engineered cover designs were provided by WDOH along with the three closure scenarios. The engineered covers considered consisted of a site soils cover, the US Ecology proposed cover, and an enhanced cover. Closure options considered included 1) ceasing waste disposal in year 2003, 2) ceasing waste disposal year 2056, and 3) ceasing waste disposal in year 2215. Disposal rates after year 2002 were assumed to be constant and given by the values in Table 2 under the column heading "Proposed". Not all covers were evaluated for each closure option. In all cases, the cover was assumed to be installed in the year 2005 over the existing trenches and infiltration in open trenches for future site operations would be controlled to no more than the cover design. Covers were assumed to begin to fail 500 years after placement in the year 2505 based on the typical cover design lifetime for Hanford Reservation facilities[2]. Cover failure was assumed to occur at this time (500 years after placement) regardless of the closure option considered. The time for the cover to degrade to natural infiltration is assumed to be the lifetime of the cover. That is, if the cover lasts 500 years, then the cover degrades to natural infiltration in 1000 years. This assumption is also based on Hanford facilities[2].

The site soils cover has a design-based infiltration rate greater than natural recharge. In this case, we have assumed that the cover does not fail, but returns to natural infiltration over time. The design-based infiltration is assumed to persist for 500 years. After that, infiltration through the cover decreases over the next 500 years and eventually returns to its natural state (0.005 m yr^{-1}) 1000 years after cover placement.

Table 7. Cover and Closure Options

Cover/closure option	Infiltration through cover ($m\ yr^{-1}$)
Site soils cover/waste disposal ceasing in 2056	0.02
Enhanced cover/ waste disposal ceasing in 2003	0.0005
Enhanced cover/ waste disposal ceasing in 2056	0.0005
Enhanced cover/ waste disposal ceasing in 2215	0.0005
US Ecology proposed cover/ waste disposal ceasing in 2056	0.002

[2] Personal communication with Michael J. Fayer, Pacific Northwest National Laboratory, Richland Washington January 24, 2003.

The compliance time is defined as the time period over which predicted doses are compared to performance objectives. WDOH issued a compliance time of 10,000 year from present. However, predicted concentrations and doses presented in this report go well beyond 10,000 years. The purpose of extending the calculations beyond 10,000 years was to understand the overall behavior of the release and transport model in light of the great uncertainty that exists in making model predictions so far in the future. Radionuclide fluxes to the aquifer were calculated out to 100,000 years and then set to zero. Radionuclide concentrations in the aquifer were calculated out to 200,000 years.

MODEL INPUT

With the exception of water fluxes and waste input rates, model input was largely taken from Rood (2000a), Rood (2000b), and Kincaid et al. (1998). Model input for radionuclide-independent parameters are presented in Table 8, and in Table 9 for radionuclide-dependent parameters. Parameters that require additional explanation and justification are discussed in separate subsections.

Table 8. Radionuclide Independent Model Input Parameters

Parameter[a]	Nominal value	Reference/Comments
Length of source parallel to groundwater flow (m)	382	Rood (2000a) (see discussion below)
Width of source perpendicular to groundwater flow (m)	518	Rood (2000a) (see discussion below)
Cover longevity (years)	500	Assumed
Source thickness (m)	10.6	Rood (2000a)
Bulk density of source (g cm^{-3})	1.97	Kincaid et al. (1998) (see discussion below)
Saturated hydraulic conductivity for source (m y^{-1})	555	Kincaid et al. (1998) (see discussion below)
van Genuchten fitting parameter α for source (m^{-1})	0.811	Kincaid et al. (1998) (see discussion below)
van Genuchten fitting parameter *n* for source	1.58	Kincaid et al. (1998) (see discussion below)
Residual moisture content for source (m^3 m^{-3})	0.015	Kincaid et al. (1998) (see discussion below)
Total porosity for source (m^3 m^{-3})	0.119	Kincaid et al. (1998) (see discussion below)
Unsaturated thickness (m)	82.3	Rood (2000a)
Number of unsaturated layers (compartments)	13	This report (see discussion below)
Thickness of each unsaturated layer (m)	6.331	This report (see discussion below)
Bulk density of unsaturated layer 1 (g cm^{-3})	1.78	Kincaid et al. (1998) (see discussion below)
Saturated hydraulic conductivity, unsaturated layer 1 (m y^{-1})	3753	Kincaid et al. (1998) (see discussion below)
van Genuchten fitting parameter α for unsaturated layer 1 (m^{-1})	1.3	Kincaid et al. (1998) (see discussion below)
van Genuchten fitting parameter *n* for unsaturated layer 1	2.1	Kincaid et al. (1998) (see discussion below)
Residual moisture content for unsaturated layer 1 (m^3 m^{-3})	0.026	Kincaid et al. (1998) (see discussion below)
Total porosity for unsaturated layer 1 (m^3 m^{-3})	0.337	Kincaid et al. (1998) (see discussion below)
Bulk density of unsaturated layers 2–13 (g cm^{-3})	1.97	Kincaid et al. (1998) (see discussion below)
Saturated hydraulic conductivity for unsaturated layer 2-13 (m y^{-1})	555	Kincaid et al. (1998) (see discussion below)
van Genuchten fitting parameter α for unsaturated layer 2-13 (m^{-1})	0.811	Kincaid et al. (1998) (see discussion below)
van Genuchten fitting parameter *n* for unsaturated layer 2-13	1.58	Kincaid et al. (1998) (see discussion below)
Residual moisture content for unsaturated layer 2-13 (m^3 m^{-3})	0.015	Kincaid et al. (1998) (see discussion below)
Total porosity for unsaturated layer 2-13 (m^3 m^{-3})	0.119	Kincaid et al. (1998) (see discussion below)
Longitudinal dispersivity in aquifer (m)	27.5	Rood (2000a)
Transverse dispersivity in aquifer (m)	5	Rood (2000a)
Well screen thickness (m)	15	Rood (2000a)
Aquifer porosity (m^3 m^{-3})	0.1	Rood (2000a)
Darcy velocity in aquifer (m y^{-1})	32.9	Rood (2000a)
Bulk density of aquifer (g cm^{-3})	1.6	Rood (2000a)

a. Time variable water fluxes and waste input rates are discussed in Water Fluxes in the Unsaturated Zone and Waste Input Rates sections later in the text.

Table 9. Radionuclide Dependent Model Input Parameters

Parameter	Nominal value[a]	Reference/Comments
Carbon K_d (mL g^{-1})	0.5	Kincaid et al. (1998)
Chlorine K_d in source (mL g^{-1})	0.75	see "Integration of Mobile Release Fraction and Partition Coefficients" section
Chlorine K_d in unsaturated zone/aquifer (mL g^{-1})	0	Kincaid et al. (1998)
Hydrogen K_d in all media (mL g^{-1})	0	Kincaid et al. (1998)
Iodine K_d (mL g^{-1})	0.5	Kincaid et al. (1998)
Protactinium K_d (mL g^{-1})	15	Kincaid et al. (1998)
Plutonium K_d (mL g^{-1})	200	Kincaid et al. (1998)
Radium K_d (mL g^{-1})	20	Kincaid et al. (1998)
Technetium K_d in source (mL g^{-1})	0.75	see "Integration of Mobile Release Fraction and Partition Coefficients" section
Technetium K_d in unsaturated zone/aquifer (mL g^{-1})	0	Kincaid et al. (1998)
Thorium K_d (mL g^{-1})	1000	Kincaid et al. (1998)
Uranium K_d (mL g^{-1})	3	Kincaid et al. (1998)
Uranium solubility (mg L^{-1})	25	Rood (2000)

a. The K_d values are for geochemical environment F as described in Kincaid et al. (1998)

Length and Width of Source

The dimensions of the source parallel and perpendicular to groundwater flow were initially taken from Rood (2000a) which had the longer side of the source oriented parallel to groundwater flow. This orientation was presumably chosen in the initial assessment because it provided a more conservative estimate of groundwater concentrations. However, further examination of head elevations presented in Kincaid et al. (1998) for the year 2100 revealed that the source should have been oriented with the long side perpendicular to groundwater flow. Therefore, the dimensions were changed in this assessment to reflect the correct orientation of the source relative to flow in the aquifer.

Length of Well Screen

The well screen length used in this assessment was based on the default value used in screening calculations at the Idaho National Engineering and Environmental Laboratory (DOE 1994). The 200-Area composite analysis (Kincaid et al. 1998) used a numerical three-dimensional aquifer flow and transport model that had vertical grid resolution of 8 m. Therefore, at a minimum, concentrations were averaged across 8 m of the aquifer. The transverse dispersivity used in the 200 Area Composite Analysis was 20 m. Presumably, the transverse dispersivity was also applied to the vertical component of dispersion in the aquifer. Using a value of 20 m for vertical dispersivity in a three-dimensional GWSCREEN simulation resulted in a significant portion of the contaminant plume extending beyond the 15-m well screen. Therefore, using a two-dimensional aquifer solution with a 15-m mixing thickness results in a conservative estimate of aquifer concentrations compared to the more realistic three-dimensional model used in the 200 Area Composite Analysis. Because it was the intent of this assessment to error on the side of

conservatism, the two-dimensional aquifer solution with a 15-m well screen was retained from the previous DEIS work and used in this assessment.

Number of Unsaturated Layers

The number of unsaturated layers or compartments in a FOLAT model simulation influences the amount of plume spreading or dispersion. Assuming a uniform compartment thickness in all compartments except the source, it was shown that a compartment thickness of 0.243 × the standard deviation of the radionuclide plume at the unsaturated-saturated interface would yield about the same amount of plume spreading as estimated by the advection dispersion equation. The standard deviation of the contaminant plume is given by

$$\sigma = \sqrt{2\alpha_L x} \tag{9}$$

where σ = the standard deviation of the contaminant plume at distance, x (m), and α_L = the longitudinal dispersivity in the unsaturated zone (m). Using the median estimated dispersivity value in Rood (2000a) of 4 m and a total unsaturated thickness of 82.3 m, we have an estimated compartment thickness of

$$T = \sqrt{2 \times 4\,\text{m} \times 82.3\,\text{m}} \times 0.243 = 6.23\,\text{m} \tag{10}$$

Dividing this value into the unsaturated thickness then provides an estimate of the number of compartments needed in the simulation (82.3 m/6.23 m = 13.199). The compartment thickness value was modified slightly to 6.331 m because the number of compartments must be a whole number. Figure 7 shows a comparison of the flux predicted by the advection dispersion equation as implemented in GWSCREEN and that produced by FOLAT. Considering the overall uncertainty in any unsaturated transport model, there is virtually no *meaningful* difference between the fluxes generated with FOLAT and GWSCREEN.

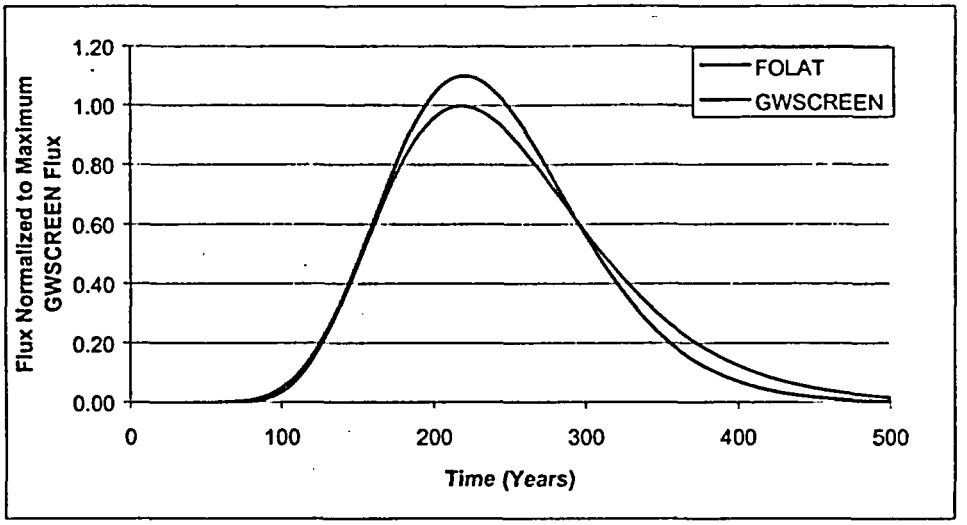

Figure 7. GWSCREEN and FOLAT flux to groundwater normalized to the maximum flux predicted by GWSCREEN for an 82.3 m unsaturated thickness and 4 m dispersivity.

Material Properties of Source and Unsaturated Zone

Material properties included bulk density, saturated hydraulic conductivity, residual moisture content, total porosity and the van Genuchten fitting parameters, α and n. The van Genuchten fitting parameters are used to determine the moisture content for a given water flux. In the original assessment (Rood 2000a), both the source and unsaturated zone had essentially the same properties. The FOLAT model allows for unique material properties assigned to the source and each unsaturated layer. Lithology of the unsaturated zone and surface soils where the trenches are located were provided in Kincaid et al (1998) for the US Ecology site (Table 10) and were used in these simulations without modification. Partition coefficients given in Table 9 are adjusted for the percent gravel content as discussed in the partition coefficient section later in the report. The aquifer was assumed to have the same percent gravel composition as unsaturated layers 2–13.

Table 10. Lithology of the Unsaturated Zone near the US Ecology Site as Described by Kincaid et al. (1998)[a]

Lithology ID	Thickness (m)	FOLAT layer	% Gravel
East Hanford Gravel	10	Source	41.7%
East Hanford Sand	6	Unsaturated 1	17.3%
Lower East Hanford Gravel	91	Unsaturated 2–13	41.7%

[a] Data from Table 4.6 page 4.82. Column ID 299-E19-1 (from Table 4.3).

Water Fluxes in the Unsaturated Zone

Water fluxes in the unsaturated zone were based on data in Gee et al. (1992), Kincaid et al. (1998), and the estimated infiltration rates for the three covers. Natural recharge in the 200 Area was estimated in Kincaid et al. (1998) to be about 0.5 cm yr^{-1}. Gee et al. (1992) estimated natural recharge to range from near zero for vegetated soils containing silt loam, to up to 10 cm yr^{-1} for unvegetated coarse sediment soils. For this assessment, the natural recharge rate is assumed to be 0.5 cm yr^{-1}. The presence of an engineered cover is assumed to limit infiltration through the waste and influence water fluxes through underlying unsaturated layers. During active disposal, a fraction of the site is excavated and water infiltration through open trenches is enhanced. Water infiltration through an open trench was assumed to be 7.5-cm yr^{-1} based on data in Kincaid et al. (1998). Closed trenches within the US Ecology property boundary during operations from 1965 to 2005 are assumed to be disturbed such that infiltration is enhanced over natural background. A value of 3-cm yr^{-1} is assumed for this time period. Because trenches are not individually modeled, infiltration across the modeled source area is area-averaged. The area-averaged infiltration rate as a function of time is given by

$$q_a(t) = q_t\,\varepsilon(t) + q_b\left(1-\varepsilon(t)\right)$$
$$\varepsilon(t) = \frac{A_o(t)}{A_T(t)} \qquad (11)$$

where

$q_a(t)$ = area average infiltration rate as a function of time (m yr^{-1}),
q_t = infiltration rate through an open trench (0.075 m yr^{-1}),
q_b = background infiltration across the site before cover emplacement (0.03 m yr^{-1}),
$\varepsilon(t)$ = the fraction of the total number of trenches that are open at time t,
$A_o(t)$ = area of active trenches at time t (m^2),
$A_T(t)$ = total area of trenches that are open or have closed at time t (m^2).

At the start of the simulation (in year 1965), water fluxes in all layers are initialized at the trench infiltration rate (7.5 cm yr^{-1}). Water fluxes in subsequent years are calculated using a preprocessor to the FOLAT program that calculates the water balance in each layer based on the user-provided water flux at the surface and the hydrologic characteristics of each soil layer. The water flux at the surface is given by Equation 11 for pre-cover times, the design-based cover infiltration rate while the cover is intact, and the background infiltration rate after the cover has failed. The cover is assumed to degrade over a period of time. The water flux through the cover after degradation begins is assumed to linearly increase from the cover designed-based infiltration rate to the natural recharge rate, over the time the cover degrades over.

Once the cover is installed, the unsaturated zone dries over time and eventually moisture contents reach an equilibrium value determined by the amount of infiltration through the cover. After cover failure, the unsaturated zone beneath the trenches is re-wetted and eventually moisture contents reach an equilibrium value determined by the natural recharge. These processes were examined first using a demonstration version of the HYDRUS 2D code (Simunek et al., 1999). The simulation used a simplified homogeneous representation of the unsaturated zone

consisting of sandy loam. Initial conditions were based on the equilibrium water contents assuming a recharge rate of 2 cm yr^{-1}. Boundary conditions included a 200 m long cover in the center of the model domain which limited infiltration to 5×10^{-4} m yr^{-1}. Free drainage was assumed at the base of the unsaturated zone. This simulation was run until water contents equilibrated throughout the domain. Equilibrium conditions were achieved after about 400 years. A second simulation was performed where the equilibrium water contents with the cover in place at 500 years were the initial conditions for the simulation. The cover was assumed to fail instantaneously and therefore, the upper boundary condition was set to a recharge rate of 2 cm yr^{-1}. The results of the two simulations are illustrated in Figure 8. The frames on the left show the advancement of the drying front following placement of the cover. The frames on the right show the advancement of the wetting front after the cover instantaneously fails.

Based on the HYDRUS simulation, the infiltration shadow beneath the composite trench appears to extend vertically down to the aquifer. Drainage from the unsaturated zone also appears to take much longer than re-wetting following cover failure.

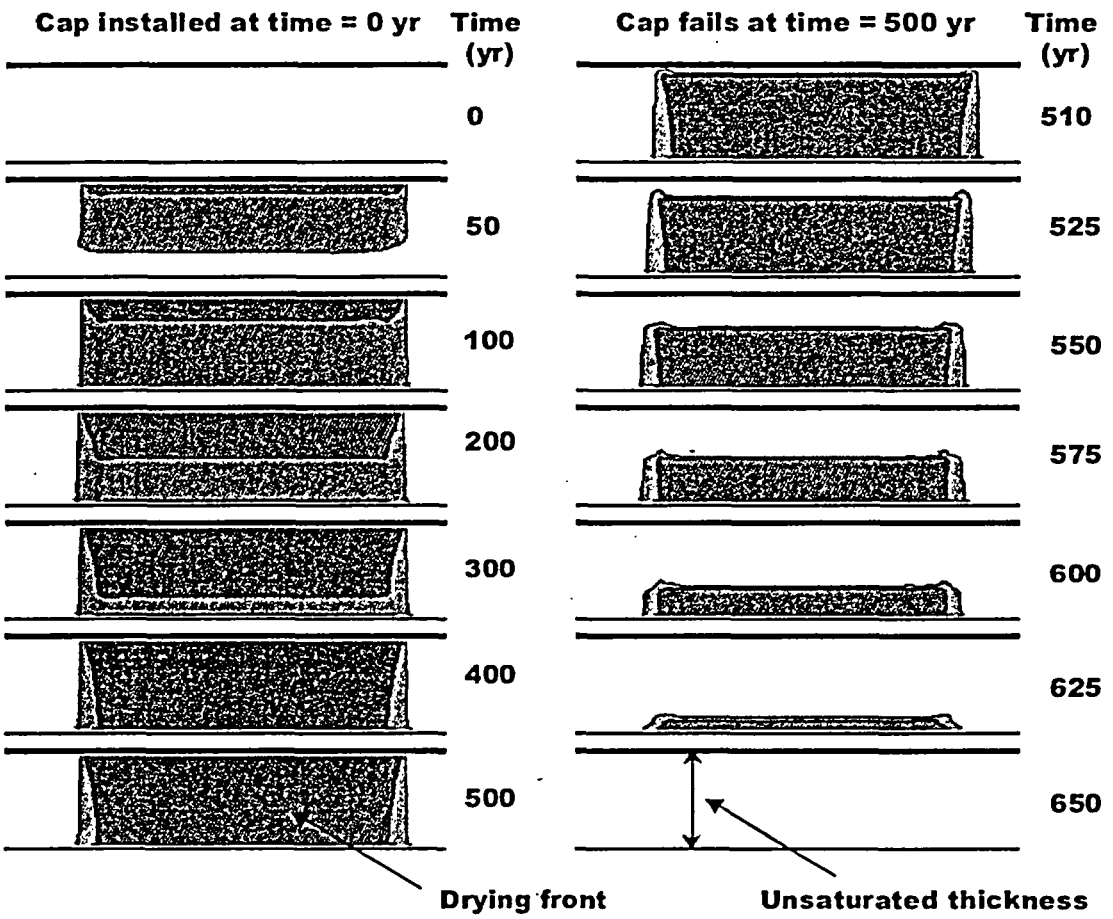

Figure 8. HYDRUS 2D simulation of moisture content profile in the unsaturated zone following cap installation in year zero (left) and cap failure in year 500 (right). Darker shades indicate drier soils. Initially, the moisture content is assumed to be constant throughout the unsaturated zone.

Installation of the cap limits infiltration into the waste and eventually throughout the "infiltration shadow". After cover failure, re-wetting of the unsaturated zone occurs relatively rapidly.

Net water flux at several depths in the unsaturated zone as calculated by the FOLAT preprocessor (FOWL) are illustrated in Figures 9 through 11 for the site soils cover, enhanced cover, and US Ecology proposed cover respectively. Note that the major effects of drying and re-wetting are incorporated into the simulation.

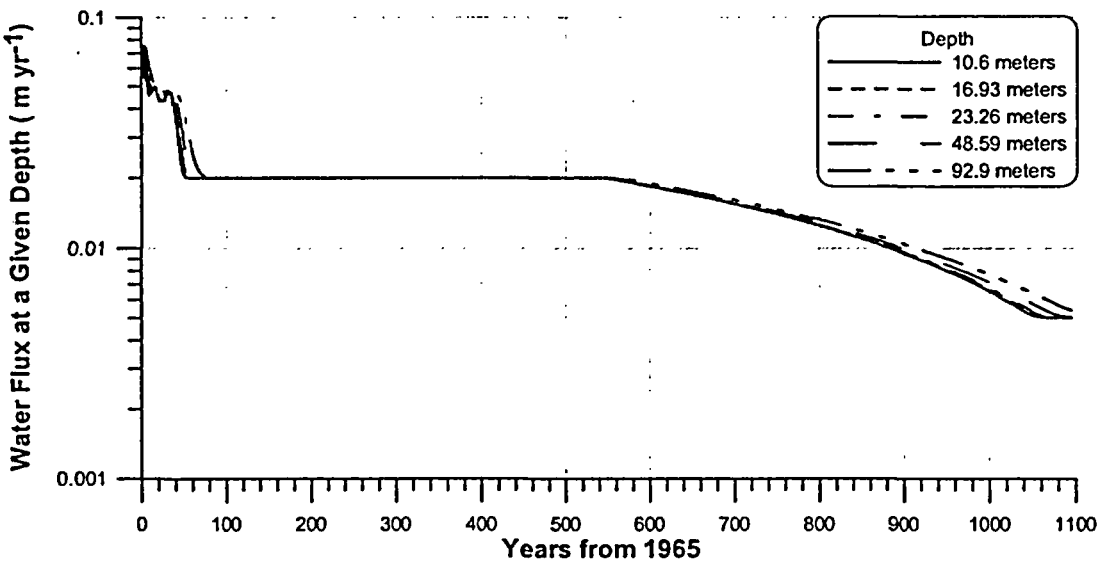

Figure 9. Water flux as a function of time for the site soils cover. Water flux through the site soil cover is 4-times natural recharge. After 500 years, water fluxes begin to return to natural recharge in each unsaturated layer. Water fluxes prior to installation of the cover in the year 2005 are controlled by the fraction of the total trench area that is open during a given year.

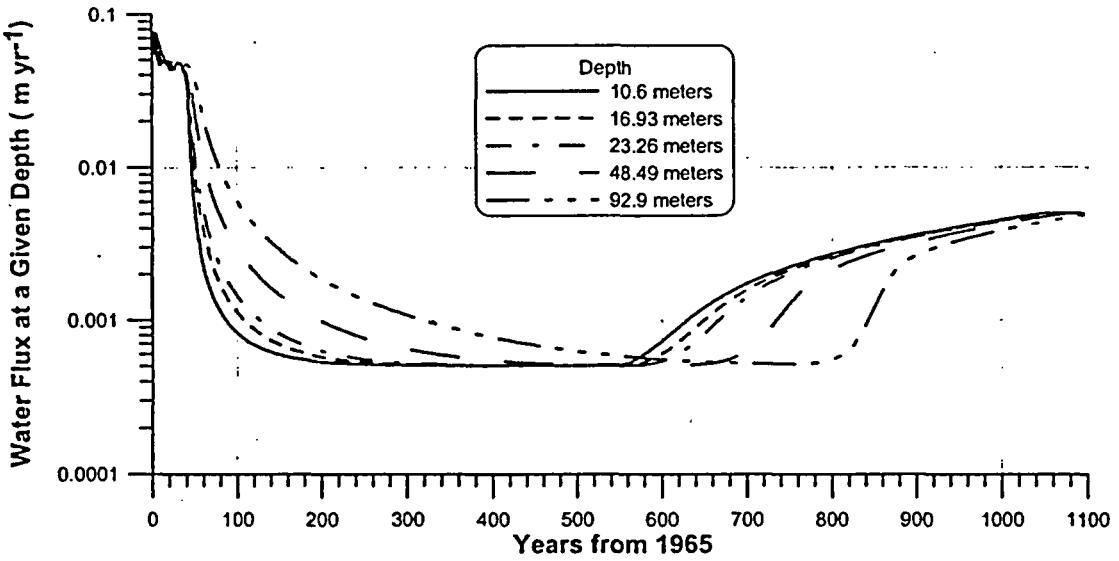

Figure 10. Water flux as a function of time for the enhanced cover. The drying front takes about 800 years to reach the aquifer. Infiltration increases beginning in year 2505, and eventually reaches the natural infiltration rate 1000 years after placement of the cover. Water fluxes prior to installation of the cover in the year 2005 are controlled by the fraction of the total trench area that is open during a given year.

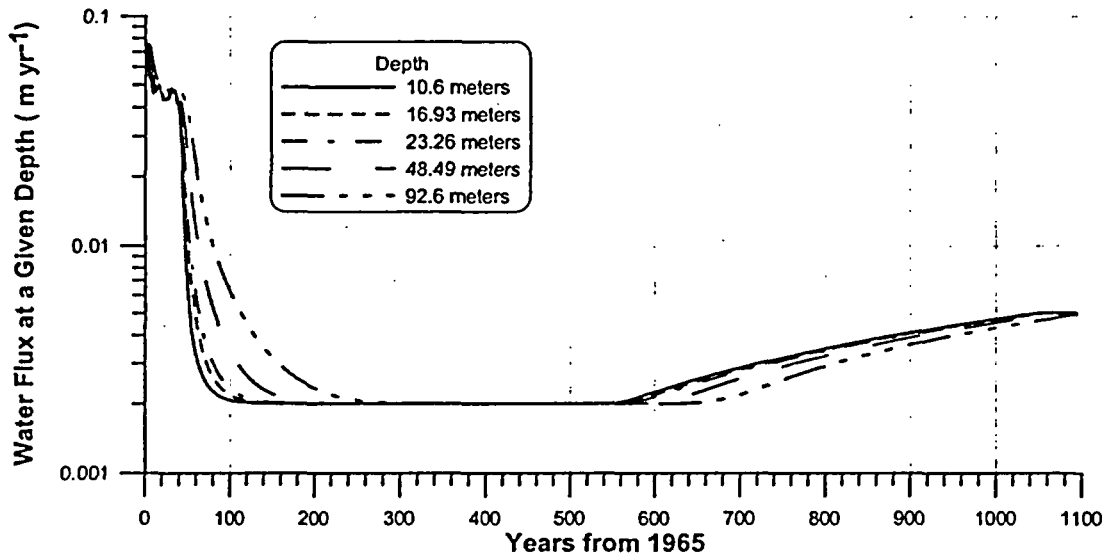

Figure 11. Water flux as a function of time for the US Ecology proposed cover. The drying front takes about 200 years to reach the aquifer. Infiltration increases beginning in year 2505, and eventually reaches the natural infiltration rate at 1000 years after placement of the cover. Water fluxes prior to installation of the cover in the year 2005 are controlled by the fraction of the total trench area that is open during a given year.

Waste Disposal Rates

Time-dependent waste disposal rates were constructed from data provided by WDOH for the radionuclides that had Phase II screening doses greater than 4 mrem yr^{-1}. For scenarios involving operation of the site beyond the year 2003, waste disposal rates were assumed to remain constant for the duration of site operations (year 2056 and year 2215). Figures 12 through 15 illustrate the disposal histories from 1965 to 2002, and the projected waste disposal rate to 2005. Figure 15 includes Ni-63 and Sr-90. Although these radionuclides were previously screened from the analysis, they have been included here because these radionuclides were detected in borehole samples taken at considerable depth below trenches. See the "Evaluation of Borehole Data" section later in this report for more details.

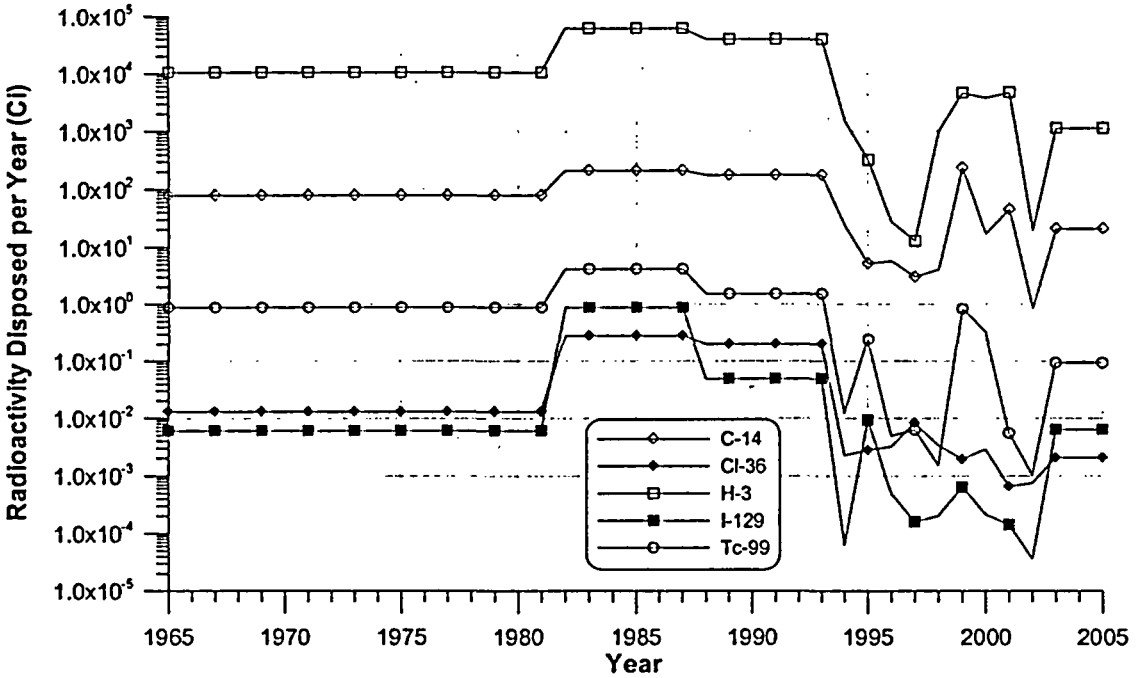

Figure 12. Radioactivity disposed in the US Ecology LLRW facility as a function of time for C-14, Cl-36, H-3, I-129, and Tc-99.

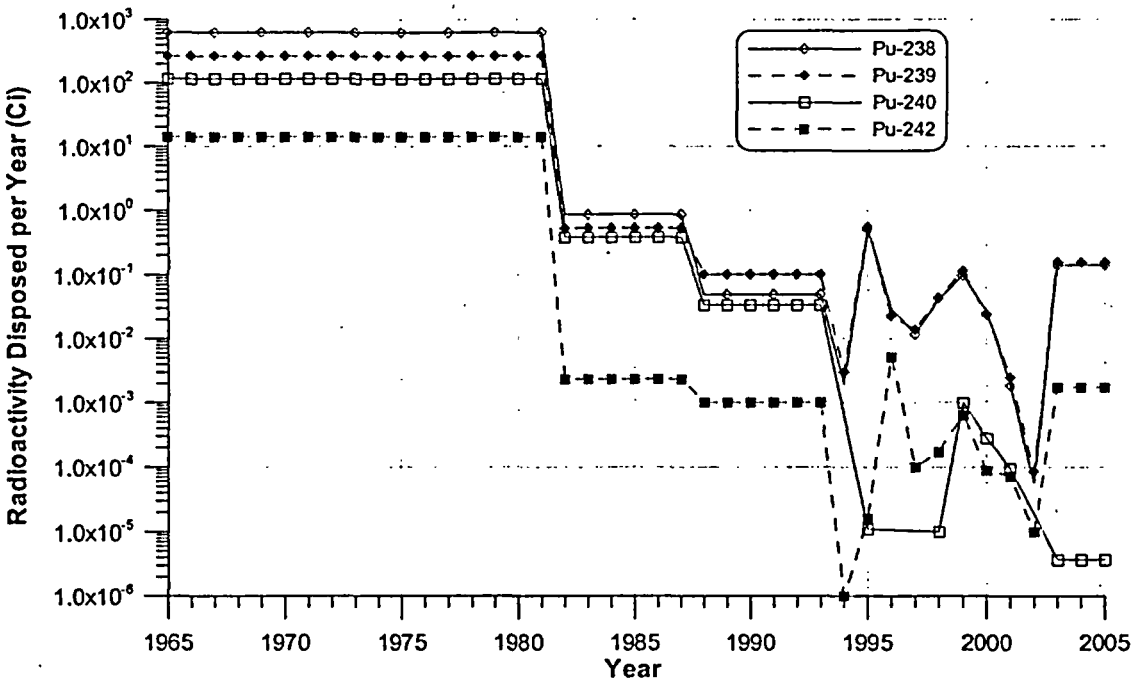

Figure 13. Radioactivity disposed in the US Ecology LLRW facility as a function of time for Pu-238, Pu-239, Pu-240, and Pu-242.

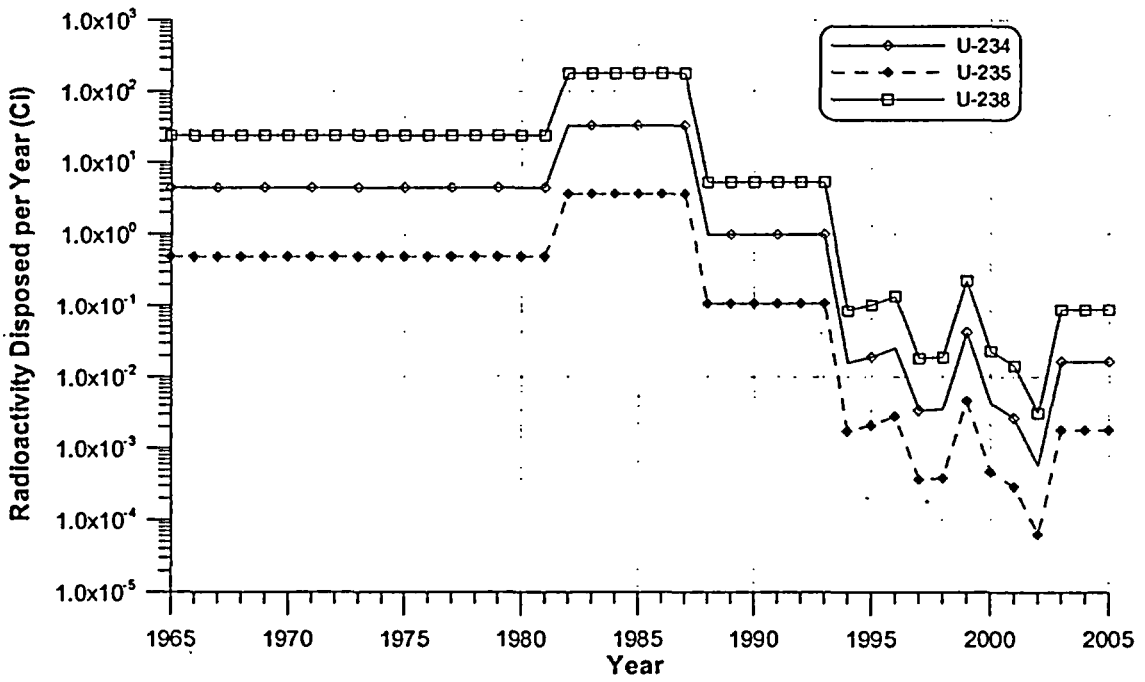

Figure 14. Radioactivity disposed in the US Ecology LLRW facility as a function of time for U-234, U-235, and U-238.

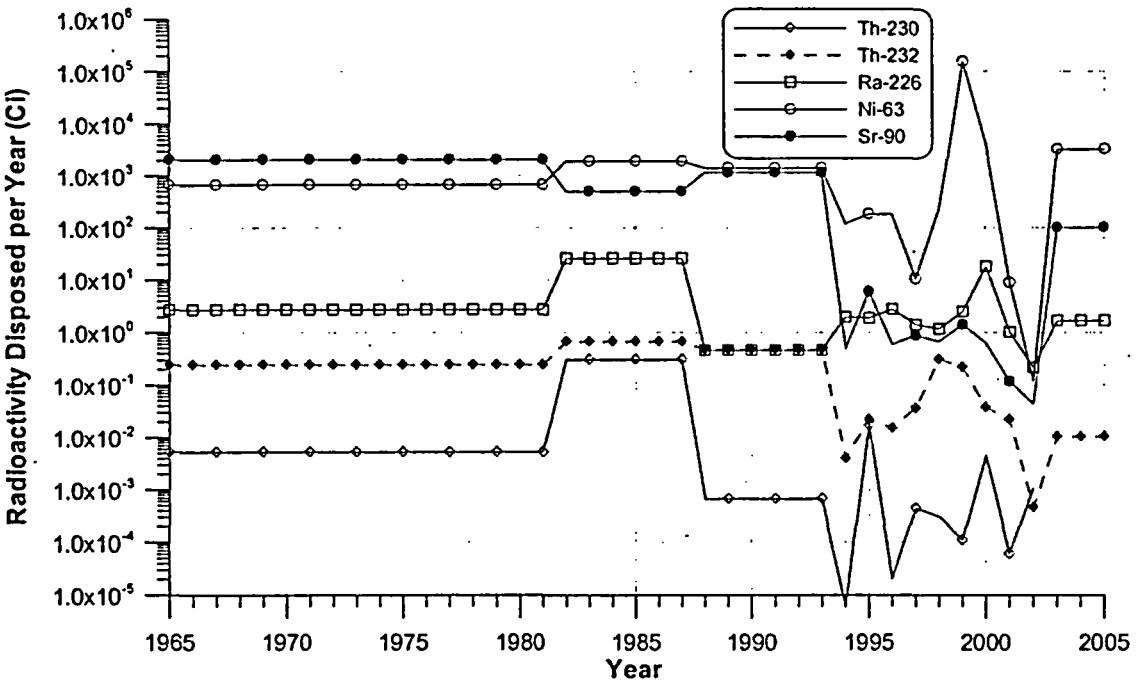

Figure 15. Radioactivity disposed in the US Ecology facility as a function of time for Th-230, Th-232, Ra-226, Ni-63, and Sr-90.

Discussion of Partitioning Coefficients

In this section, we first review the partitioning coefficients that were used in the 200 Area Composite Analysis (Kincaid et al 1998). We then qualitatively evaluate the partitioning coefficients used for the US Ecology site in the 200 Area Composite Analysis (Kincaid et al. 1998) in light of recent measurements of radionuclides in boreholes beneath trench 5 and introduce a transport model that may explain the radionuclide measurements below trench 5.

The 200 Area Composite Analysis (Kincaid et al. 1998) provided estimates of element-specific K_d values for six different geochemical environments identified as A through F. In general, the A environment (described as high organic and very acidic) had the lowest K_d values and the F environment (described as low organic, low salt, and near neutral) had the highest K_d values. Geochemical environments were then assigned to three zone categories; high impact, intermediate impact, and low impact/groundwater.

The high impact zone category was defined as the area in the unsaturated zone near the source that is impacted by the chemical composition of the waste, particularly any contaminated liquids that were disposed. Organic compounds, pH, and salt, when present in the source may affect the K_d values. The high impact zone category has the lowest K_d values

The intermediate impact zone category was assigned to the unsaturated zone where the excessive acidic or basic nature of the waste has been neutralized by the buffering capacity of the natural soil and no pH effects of the plume remain.

The low impact/groundwater zone category was defined in the unsaturated zone and unconfined aquifer where K_d values are not affected by the chemical composition of the

contaminant plume. The chemical properties of the waste are assumed to be so greatly diluted that they do not affect the K_d value. The groundwater zone category has the highest K_d values.

In the 200 Area composite analysis (Kincaid et al. 1998), the US Ecology site was assigned a geochemical environment described as low organic/low salt/near neutral (geochemical environment F) for all soils in the unsaturated zone and aquifer. Consequently, the same K_d values were assigned to all geologic media (although K_d values were modified for the percent gravel in the lithology).

The K_d values used for the US Ecology site in the 200 Area Composite Analysis (Kincaid et al. 1998) are relatively high; for example, the nickel K_d was estimated to be 300 mL g^{-1}. A K_d of this magnitude for nickel would result in little present-day migration of nickel to the unsaturated zone and is at odds with recent measurements of Ni-63 in boreholes beneath Trench 5.

It is beyond the scope of this assessment to fully investigate the mobility of each radionuclide of interest. Radionuclide transport may be a function of many other processes such as colloid transport, presence of complexing agents, and preferential flow paths. In the next section, we examine the borehole data and propose a mobile-fraction transport model that may explain the observed distributions of radioactivity with depth for the radionuclides detected. The mobile-fraction transport model separates radionuclide inventories into a mobile and immobile fraction. The mobile fraction is then calibrated to the measured borehole data for the radionuclides where measurements are available. For the immobile fraction, we have used the K_d values reported in Kincaid et al. (1998) for geochemical environment F without modification. Partition coefficients reported in Table 9 were later modified by the percent gravel in the rock matrix because sorption was assumed to take place only on the fine material and not the course gravel component. The partition coefficient adjusted for gravel content is given by

$$K_d(adjusted) = K_d \times (1 - f_g) \tag{12}$$

where f_g is the fraction of gravel in the rock matrix.

Evaluation of Borehole Data

In 1999, US Ecology conducted a comprehensive facility investigation (US. Ecology 1999). Part of the investigation was to examine radionuclide migration from the disposal trenches, which entailed the drilling of four boreholes to a depth of about 21.3 m (70 feet) below the trench bottom. Two boreholes were drilled adjacent trench 5 (borehole C and D) and two adjacent the chemical disposal trench (borehole A and B). Radionuclides were measured as a function of depth below the boreholes and included Ni-63, Sr-90, Tc-99, Pu-238,239/240, U-234,235,238, Th-230,232, and Ra-226 (Appendix A).

Nickel-63 and Sr-90 had soils concentrations above the minimum detectable concentration (MDC) in almost all the samples and showed relatively uniform concentration with depth. These results included the samples taken beneath chemical trench, which presumably received no radionuclides. This distribution reflects relatively rapid transit times in the unsaturated zone. In fact, to produce the observed depth distribution, the radionuclides would have to been traveling with infiltrating water with essentially no sorption on the rock matrix, which is at odds with laboratory data on the mobility of strontium and nickel.

Plutonium had several samples with plutonium concentrations at or above the MDC. These occurred only in samples taken below the chemical disposal trench.

Uranium, thorium and radium isotopes all had soil concentrations above the MDC. Minor (2001) reviewed these data and concluded "*...the analytical results of nine vadose zone samples agree well with local background concentrations and/or represent data whose quality appear to be reliable—potassium-40; 226- and 228-radium; 228-, 230-, and 232-thorium; and 234-, 235-, and 238-uranium.*" Analysis of uranium isotopic ratios in the borehole samples however suggested that the some of the uranium detected was anthropogenic in nature. If the uranium measured in the boreholes represented naturally occurring uranium in soils, then we would expect the relative proportions of U-238, U-235, and U-234 to be close to their natural abundance (Table 11). However, the mean U-235 weight percent in borehole samples (Table 12) was substantially higher than that for natural uranium, and in fact, was closer to that of enriched uranium. (see Appendix A). Assuming a U-235 enrichment of 3% by weight, the weight percent of U-234 also increases from 0.0054% to 0.017% based on the empirical relationship proved in Bowman and Suto (1996). Although the measured U-235 weight percent appears to suggest an enriched source of uranium, the U-234 weight percent is close to what would be expected from natural uranium. Of the 37 samples analyzed for U-235, 26 were above the MDC (compared to all U-238 and U-234 samples), suggesting greater uncertainty in the U-235 measurement. It is unknown whether a systematic positive bias existed in the U-235 sample analysis. Another interesting observation is that U-238 concentrations in borehole B are substantially lower than those in the other boreholes. If the uranium were from natural sources, then we would expect uranium concentrations in all boreholes to be about the same. It is beyond the scope of this assessment to examine this issue any further. For the purpose of model calibration, we have assumed the U-238 detected in borehole B represents natural sources and subtracted the depth-averaged value (0.048 pCi g^{-1}) from values in boreholes C and D. The net U-238 concentrations were then assumed to be derived entirely from the waste disposed. Calibration was not performed for U-235 and U-234.

Table 11. Properties of Uranium-234, -235, and -238 for One Mole of Natural Uranium.

Isotope	Specific Activity (Ci g^{-1})	% isotopic abundance	Mass (g)	Activity (Ci)	% weight abundance
U-238	3.35172E-07	99.2745%	236.27	7.9192E-05	99.2836%
U-235	1.90291E-06	0.7200%	1.692	3.2197E-06	0.7110%
U-234	6.24393E-03	0.0055%	0.01287	8.0359E-05	0.0054%

Table 12. Statistics of the Distribution of U-238, U-235 and U-234 Percent Weight Abundance in Bore Hole Samples

Statistic	U-238 % weight abundance	U-235 % weight abundance	U-234 % weight abundance
Mean	95.9209%	4.0734%	0.0057%
Standard Deviation	2.6592%	2.6590%	0.0014%
Minimum	90.3091%	1.0347%	0.0042%
Maximum	98.9591%	9.6862%	0.0106%
Number of observations	26	26	26

Release and Transport Model Simulations of Trench 5

The measured concentrations in boreholes C and D (taken below trench 5) and the estimated radioactivity disposed of in trench 5 provide the necessary data to construct a release and transport model of the trench. The chemical trench presumably received no radionuclides and therefore, there is no estimate of the amount of radioactivity that the trench received. For this reason the chemical trench was not modeled. The model considers radioactive waste disposed in the trench in the years 1978–1979. During active disposal, the trench is open and there is no runoff. An infiltration rate of 7.5 cm yr^{-1} is assumed during active disposal, consistent with the estimate of Kincaid et al (1998). During the period the trench was open (April 1978 to September 1979), 22.9 cm of precipitation was recorded at Pasco according to precipitation records obtained from the National Climatic Data Center, so the assumed infiltration rate during this period is reasonable. After closure of the trench, infiltration is assumed to be reduced to 3 cm yr^{-1}. Radionuclide concentration profiles below the trench suggest the radionuclides are moving with the infiltrating water with little or no sorption. To account for the observed radionuclide distribution in soil, the proposed model assumes that there is a small, but mobile fraction of radionuclides in the waste. This fraction, referred to as the mobile fraction hereafter, is easily leached and moves with the infiltrating water. The mechanism for movement could be colloidal transport or chemical complexation with chemicals that may have been disposed of in the trench. However, the model is empirical in nature and does not attempt address the mechanisms of release or transport.

The viability of the proposed model is evaluated by comparing the activity below the trench to the estimated activity in the trench. Assuming the soil concentrations are horizontally uniform across the area beneath the trench, the amount of activity that is below the trench (to a depth of 21.3 m below the bottom of the trench) can be estimated by numeric integration. The integrated activity is given by

$$Q = A\rho \int_0^b C(x)\,dx \qquad (13)$$

where

Q = integrated activity from the bottom of the trench to depth b (Ci)
A = area of the trench (m^2)
$C(x)$ = soil concentration as a function of depth (Ci g^{-1})
ρ = bulk density of soil (1.9×10^6 g m^{-3})
b = depth below trench (m)

The function, $C(x)$ was generated by averaging the soil concentrations in borehole C and D at each depth. The activity disposed of in trench 5 was estimated from the total radioactivity disposed from 1965 to 1981 reported by WDOH (Table 13). It was assumed that each trench that was open during the 1965 to 1981 time frame received an equal amount of radioactivity. Seven trenches were operating during this time, therefore, the estimated activity disposed in trench 5 was the total 1965 to 1981 disposed radioactivity divided by seven. Measured concentrations of Pu-239 and Pu-240 were not segregated and reported as single value because it is almost

impossible to resolve the two isotopes using alpha spectroscopy. The primary alpha decay energy for Pu-239 is 5.156 MeV (73.1%) and 5.168 MeV (73%) for Pu-240. Therefore, inventories of Pu-239 and Pu-240 were summed.

Table 13. Estimated Radionuclide Inventories Disposed in Trench 5 and Integrated Radionuclide Radioactivity below Trench 5 to a Depth of 21.3 m Below the Bottom of the Trench

	Ni-63	Sr-90	Tc-99	Pu-239/240	U-238	Pu-240	Pu-239
Total 1965-81 inventory (Ci)	1.127E+04	3.462E+04	1.481E+01	6.444E+03	4.054E+02	1.949E+03	4.494E+03
Estimated inventory in trench 5 (Ci)	1.610E+03	4.946E+03	2.116E+00	9.205E+02	5.791E+01	2.785E+02	6.420E+02
Integrated radioactivity to 21.3 m below trench 5 (Ci)	4.932E-01	3.428E-02	6.322E-02	2.379E-03	1.162E-02	n/a	n/a

The integrated radioactivity below the trench may not represent the total radioactivity released from the trench. The mobile-fraction model was used to determine the total activity released from the trench by calibrating the mobile fraction inventory to the distribution of soil concentrations below the trench.

The objective of the calibration was to match radionuclide concentrations in the borehole samples taken below trench 5 to the model-estimated concentrations in unsaturated layers 1–3 which lie at a depth between 10.6 m to 29.5 m (35 ft to 96.8 ft) below ground surface. Measured concentrations were averaged across the thickness of each unsaturated layer. The three unsaturated layers correspond to following depths below the ground surface: 10.6 to 16.9 m for unsaturated layer 1, 16.9 to 23.26 m for unsaturated layer 2, and 23.26 to 29.6 m for unsaturated layer 3. Measured concentrations that were below the minimum detectable concentration (MDC) were assumed to be equivalent to the MDC for this calculation, which provides a conservative estimate of the radioactivity below the trench. The FOLAT model outputs radionuclide pore water concentrations and moisture contents in each layer as a function of time. Radionuclide pore water concentrations in each layer were converted to radioactivity per unit mass (pCi g^{-1}) using

$$C_m = \frac{\theta C_w \left(1 + \frac{K_d \rho}{\theta}\right)}{\rho} \tag{14}$$

where

C_m = radionuclide concentration per unit mass of soil (pCi g^{-1})
C_w = radionuclide concentration in pore water (pCi cm^{-3})
θ = moisture content (cm^3 cm^{-3})
ρ = bulk density of soil (g cm^{-3}).

For comparison, the DUST model (Sullivan 1996) was also run in parallel with FOLAT. The DUST model uses a finite-difference approximation to the advection-dispersion equation to solve for concentrations in the unsaturated zone and provides a verification of the FOLAT model output. Additionally, DUST outputs concentrations on a finer scale than FOLAT, which was important for illustrating the relative migration of the mobile and immobile fraction in the

unsaturated zone. However, the finer scale was not important for calculating flux to the aquifer and for this and other reasons stated earlier, DUST was not utilized in the overall assessment model.

The primary calibration objective was to minimize the bias in the average predicted concentrations over the sampling depth, although a slight positive bias (indicating model overprediction) was considered acceptable. A second calibration objective was to minimize the residuals between the predicted and observed concentrations. Calibration objectives were achieved by adjusting the fraction of the radionuclide inventory that is considered mobile until the calibration objective was met. Model calibration was based only on the FOLAT simulations.

The metrics used to evaluate model calibration incorporate several performance measures commonly used in evaluation of atmospheric transport models (Fox 1981; EPA 1988; Cox and Tikvart 1990). These measures were the fractional bias (*FB*) and normalized mean square error (*NMSE*). The *FB* was given by

$$FB = \frac{2\left(\overline{C_o} - \overline{C_p}\right)}{\left(\overline{C_o} + \overline{C_p}\right)} \tag{15}$$

where C_p and C_o were the predicted and observed concentrations, respectively. Overbars indicated averages over the sample. The *NMSE* was given by

$$NMSE = \frac{\overline{\left(C_o - C_p\right)^2}}{\overline{C_o}\,\overline{C_p}} \tag{16}$$

The *FB* is a measure of the mean bias. A *FB* of 0.67 is equivalent to model under prediction by a factor of 2. A negative value indicates model over prediction. A *FB* value of ±0.3 indicates model bias is roughly ±25%. That is, model predictions are either over- or under predicted by factor of 1.35.

The *NMSE* is a measure of model variance. A *NMSE* value of 1.0 indicates that the typical difference between predictions and observations is approximately equal to the mean. A perfect model would have a *FB* and *NMSE* of zero. Our calibration targets for *FB* and *NMSE* were abs(*FB*) ≤ 0.1 and *NMSE* ≤ 0.1, although these targets were not met in all cases. Excursions of abs(*FB*) above 0.1 were acceptable is the *FB* was negative, indicating model over prediction.

Results of the calibration (Table 14) indicate that all calibration objectives were met. Figures 16 and 17 illustrate the measured and model predicted concentrations as a function of depth below the trench for Ni-63 and Sr-90. The measured data shown are the average concentration in borehole C and D at each sampling depth. The DUST simulation includes both the mobile and immobile fractions. The immobile fraction only migrates about a meter below the bottom of the trench whereas the mobile fraction extends beneath the 21.3 m sampling depth from the bottom of the trench. The measured data in Table 14 are the layer-averaged concentrations. Release fractions were also calculated and are consistent with what we might expect. Technetium-99 had the highest release fraction, which might be expected because it has a low capacity for sorption and is relatively mobile in the environment. It is suspected that the Tc-99 concentrations probably reflect some dissolved-phase transport with some partitioning occurring

in the waste form. However, it is important to note that if we assumed the *entire* Tc-99 inventory moved with the infiltrating water, then the model-predicted concentrations in the unsaturated zone would be much greater than what was observed.

Table 14. Results of Model Calibration to Trench 5 Measurement Data using FOLAT

	Ni-63	Sr-90	Tc-99	Pu-239/240	U-238[d]
Measured concentration in unsaturated layer 1 (pCi g^{-1})[a]	5.48E+00	1.60E-01	5.67E-01	2.45E-02	9.50E-02
Measured concentration in unsaturated layer 2 (pCi g^{-1})[b]	4.73E+00	8.65E-02	6.12E-01	2.53E-02	9.15E-02
Measured concentration in unsaturated layer 3 (pCi g^{-1})[c]	5.07E+00	1.44E-01	5.72E-01	2.33E-02	1.22E-01
Predicted concentration in unsaturated layer 1 (pCi g^{-1})	6.60E+00	1.65E-01	7.58E-01	3.18E-02	1.02E-01
Predicted concentration in unsaturated layer 2 (pCi g^{-1})	5.30E+00	1.32E-01	6.09E-01	2.55E-02	1.06E-01
Predicted concentration in unsaturated layer 3 (pCi g^{-1})	3.94E+00	9.85E-02	4.53E-01	1.90E-02	1.07E-01
Fractional bias	-3.59E-02	-1.22E-02	-3.81E-02	-4.51E-02	-2.17E-02
Normalized mean square error	3.51E-02	8.16E-02	4.77E-02	3.84E-02	1.47E-02
Calibrated mobile release quantity (Ci)	1.00E+00	3.50E-02	1.00E-01	4.20E-03	2.00E-02
Calibrated mobile fraction	6.21E-04	7.08E-06	4.73E-02	4.56E-06	3.45E-04

a. Average of samples taken between 0, 2.4, and 5.2 m below bottom of trench.

b. Average of samples taken between 7.9 and 10.7 m below bottom of trench

c. Average of samples taken between 13.4 and 21.3 m below bottom of trench

d. A background value of 0.048 pCi g^{-1} was subtracted from the measured concentration.

The other radionuclide (Ni-63, Sr-90, Pu-239/240, and U-238) exhibit much lower release fractions. These nuclides are known to sorb and would move little in the 20-year period (1979–1999) if only dissolved-phase transport were considered. The presence of organic matter and acidic conditions may enhance dissolved-phase transport; however, total organic carbon measurements below the trench were typically less than 1000 mg kg^{-1} or <0.1% by weight, which is on the lower end of the distribution of organic carbon contents observed in soils (Lyman et al. 1982). Release fractions for plutonium and strontium were similar, but uranium and nickel were about 2 orders of magnitude greater. The release fraction calculation is sensitive to the estimated initial inventory. Evaluation of the uncertainty in the inventory estimate was beyond the scope of this project, but is recommended for future work.

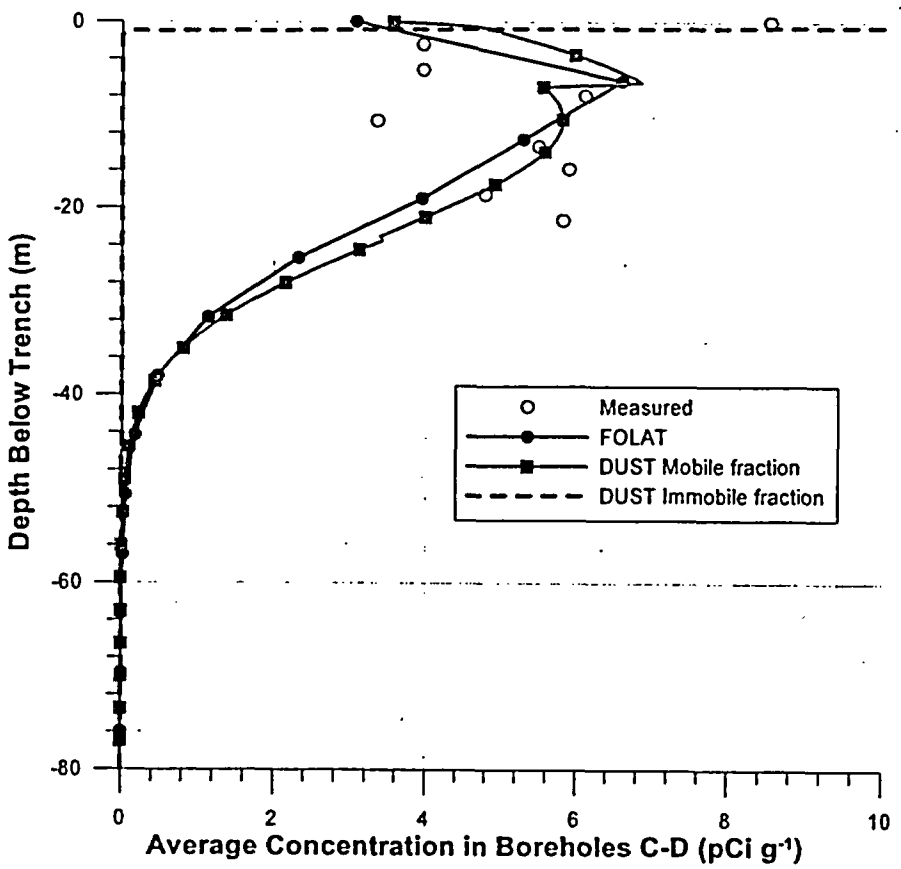

Figure 16. Predicted and observed Ni-63 soil concentrations below trench 5. The concentrations predicted with DUST include the mobile and immobile fraction. The immobile fraction was calculated using the K_d values for geochemical environment F as described in Kincaid et al. (1998). Measured concentrations represent the average between boreholes C and D at each sampling depth.

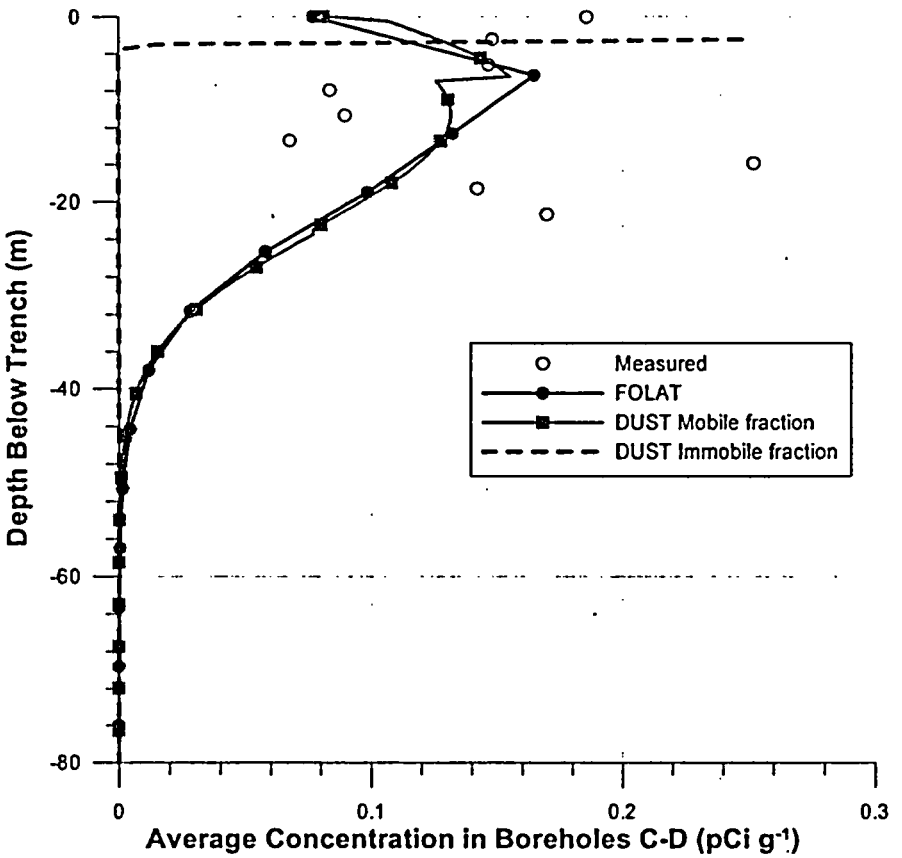

Figure 17. Predicted and observed Sr-90 soil concentrations below trench 5. The concentrations predicted with DUST include the mobile and immobile fraction. The immobile fraction was calculated using the K_d values for geochemical environment F as described in Kincaid et al. (1998). Measured concentrations represent the average between boreholes C and D at each sampling depth.

Colloid transport appears to be a viable mechanism to explain the observed distribution of radionuclides with depth below the trench. This mechanism involves either the physical movement of colloid-sized (0.1 – 1 μm) particles of the radionuclide itself, or physical movement of a radionuclide attached to a colloidal-sized soil particle. Colloids will move with the infiltrating water until they are physically trapped within the rock matrix. Additionally, water fluxes may need to reach some minimum threshold in order for the colloid to move. For this assessment, we have assumed that colloids behave as a dissolved substance with no sorption and move with the velocity of infiltrating water. This assumption provides a conservative estimate of radionuclide flux to the aquifer because colloids are assumed to never be physically trapped within the rock matrix, and there is no water flux threshold for their movement. Colloid transport is currently an area of ongoing research and it is beyond the scope of this assessment to investigate this transport mechanism any further. However, the mobile-fraction model employed provides radionuclide concentrations in the unsaturated zone that are consistent with measured data and provides conservative estimates of radionuclide fluxes to the groundwater.

Integration of the Mobile Release Fraction and Partition Coefficients

Based on the results of the mobile-fraction model calibration, a fraction of each radionuclide in the inventory (except H-3, Tc-99 and Cl-36) was assumed to be mobile. The fraction was only applied to the 1965-2002 inventory. Future disposals were assumed to be controlled so as to minimize mobile-fraction releases. The mobile fraction was based on assumed similarity to other isotopes and/or sorption characteristics. Mobile release fractions were assigned as follows

- A mobile release fraction of 3.45×10^{-4} was assigned to all uranium isotopes
- A mobile release fraction of 6.21×10^{-4} was assigned to I-129 and C-14
- A mobile release fraction of 4.56×10^{-6} was assigned to all plutonium, thorium and radium isotopes
- A mobile release fraction of 1.0 was assigned to tritium

Iodine-129 and C-14 were assumed to have the same mobile release fraction as Ni-63 not because these radionuclides are chemically similar to nickel, but because nickel had the highest mobile release fraction (with the exception of Tc-99, see discussion in next paragraph). Tritium can move both in a dissolved phase and vapor phase. Vapor phase transport would likely result in a substantial quantity of H-3 released to the atmosphere. For this groundwater assessment, we have conservatively assumed all the tritium moves down with infiltrating water.

A mobile release fraction for Tc-99 is more complicated, because some of the Tc-99 detected in the unsaturated zone was probably from dissolved-phase transport. For Tc-99 and the other mobile radionuclide (Cl-36) an "effective" K_d in the source was calculated by calibrating the predicted Tc-99 integrated activity from the bottom of the trench to a depth of 21.3 m below the trench to the corresponding integrated measured activity using the *total* activity disposed of in trench 5. This "effective" K_d represents partitioning from the waste form into infiltrating water. If we were to apply the nominal K_d value of 0 mg L^{-1} to the entire Tc-99 inventory in the trench, then concentrations in the unsaturated zone would be grossly overpredicted. Recall that only 4.73% of the Tc-99 inventory was estimated to have left trench 5. Because the nominal K_d value in the unsaturated zone is zero for Tc-99 and Cl-36, transport times will be the same as the mobile release fractions.

For the remainder of the radionuclides (excluding Tc-99, Cl-36, and H-3), immobile fraction leaching from the trench and transport in the unsaturated zone and aquifer used partition coefficients for geochemical environment F provided in Kincaid et al. (1998).

Both Ni-63 and Sr-90 were eliminated from further consideration in the Phase II screening. To evaluate the radiological dose potential for the mobile fraction of these radionuclides, an additional screening exercise was performed. The Ni-63 and Sr-90 mobile fraction was multiplied by the 1965–2215 inventory (877,535 Ci for Ni-63 and 65,688 Ci for Sr-90) and the entire mobile fraction inventory was assumed to be placed instantaneously in trench 5. A GWSCREEN simulation was run using the Phase II screening infiltration and transport parameters. A receptor well was placed on the downgradient edge of the trench. The maximum doses for Ni-63 and Sr-90 were 0.037 mrem yr^{-1} and 0.021 mrem yr^{-1} respectively. Because these doses were below the screening cutoff of 4 mrem yr^{-1}, further consideration of the nuclides was not warranted.

DETERMINISTIC AQUIFER FLUXES, CONCENTRATIONS, AND DOSES

In this section, integrated radionuclide fluxes to the aquifer, aquifer concentrations, and drinking water doses are presented. Drinking water doses are calculated assuming 2 liters of water are ingested per day for 365 days per year and using the ICRP ingestion dose conversion factors presented earlier in Table 6. Because of the large volume of output generated by the transport model, integrated radionuclide fluxes to the aquifer and groundwater concentrations are only summarized in tables. Detailed output is available electronically via Microsoft Excel spreadsheets and the raw ASCII output for each of the radionuclides analyzed. Doses are presented graphically for each cover/closure scenario. Results are presented on a time scale that begins in the year 1965, the year the US Ecology began operations.

Radionuclide Fluxes to the Aquifer

Radionuclide fluxes to the aquifer were numerically integrated from zero to 10,000 years using a Simpson Rule integration routine (Press et al. 1992) and the unsaturated-saturated radionuclide fluxes generated by the FOLAT model (Table 15). Each FOLAT simulation was truncated at 100,000 years and radionuclide fluxes to the groundwater were set to zero after this time. The integrated fluxes provide a measure of the overall source-unsaturated zone mass balance and the relative merits of each of the cover designs.

Cover design had a minor impact on the integrated radionuclide flux to the aquifer for long-lived mobile radionuclides (Cl-36, Tc-99) and the mobile fraction of relatively immobile radionuclides. However, groundwater concentrations are also influenced by the rate of radionuclide release, which is a function of the infiltration rate through cover. Higher infiltration rates result in higher radionuclide fluxes to the aquifer and higher radionuclide concentrations in the aquifer. The integrated flux to the groundwater may be the same for low and high infiltration rates.

Integrated radionuclide fluxes to the aquifer for relatively short-lived radionuclides (C-14 and H-3) exhibit greater sensitivity to cover design. For example, there is factor of 2.6 decrease in the integrated H-3 flux between the enhanced cover and site soils cover. Cover design influences the 0–10,000 year integrated radionuclide aquifer flux for the immobile actinide fraction primarily by delaying the arrival time in the aquifer. For some of the shorter-lived plutonium and thorium isotopes (Pu-238 and Th-230) and Ra-226, little of the total immobile radionuclide inventory ever reaches the aquifer because of radioactive decay.

There appears to be a discrepancy in the fraction released to groundwater for the immobile fraction of the uranium isotopes. The fraction released to groundwater for the U-234 and U-235 immobile fractions are about the same, but the corresponding U-238 value is substantially smaller. The reason for this is the U-238 release is solubility limited whereas U-234 and U-235 releases are not. Because the specific activity of U-238 is much smaller than U-234 or U-235, and there is much more U-238 disposed, the uranium solubility controls the release of U-238 from the source.

Table 15. Zero to 10,000 year Integrated Groundwater Fluxes for the Three Cover Designs and Year 2056 Closure Date

Radionuclide[a]	Site soils cover (Ci)	Fraction released to aquifer	Enhanced cover (Ci)	Fraction released to aquifer	Proposed cover (Ci)	Fraction released to aquifer
H-3	1.82E+03	2.12E-03	7.00E+02	8.14E-04	7.12E+02	8.28E-04
C-14	1.56E+03	3.07E-01	5.42E+02	1.07E-01	6.09E+02	1.20E-01
C-14MF	2.45E+00	9.80E-01	2.19E+00	7.86E-01	2.26E+00	8.11E-01
Cl-36	3.22E+00	9.98E-01	3.08E+00	9.55E-01	3.10E+00	9.60E-01
I-129	4.22E+00	7.07E-01	1.76E+00	2.95E-01	1.96E+00	3.28E-01
I-129MF	3.51E-03	1.00E+00	3.51E-03	9.93E-01	3.51E-03	9.93E-01
Tc-99	5.36E+01	9.73E-01	5.12E+01	9.30E-01	5.15E+01	9.35E-01
U-238	1.23E-04	8.14E-08	3.80E-06	2.52E-09	5.22E-06	3.46E-09
U-238MF	5.17E-01	1.00E+00	5.16E-01	9.89E-01	5.16E-01	9.90E-01
U-235	4.63E-05	1.51E-06	1.40E-06	4.59E-08	1.93E-06	6.31E-08
U-235MF	1.05E-02	1.00E+00	1.05E-02	9.90E-01	1.05E-02	9.90E-01
U-234	4.13E-04	1.48E-06	1.25E-05	4.47E-08	1.72E-05	6.15E-08
U-234MF	9.56E-02	1.00E+00	9.53E-02	9.87E-01	9.54E-02	9.88E-01
Th-230	2.03E-40	1.04E-40	3.09E-42	1.58E-42	4.48E-42	2.30E-42
Th-230MF	8.87E-06	9.99E-01	8.79E-06	9.76E-01	8.82E-06	9.79E-01
Ra-226	2.67E-16	8.26E-19	3.99E-18	1.23E-20	5.80E-18	1.79E-20
Ra-226MF	9.86E-04	9.31E-01	6.62E-04	5.49E-01	7.40E-04	6.13E-01
Pu-238	2.71E-37	2.55E-41	3.96E-45	3.73E-49	2.15E-44	2.03E-48
Pu-238MF	1.56E-02	3.33E-01	3.00E-03	6.20E-02	3.53E-03	7.31E-02
Pu-239	2.42E-27	5.36E-31	3.84E-29	8.53E-33	5.55E-29	1.23E-32
Pu-239MF	1.98E-02	9.96E-01	1.94E-02	9.44E-01	1.95E-02	9.51E-01
Pu-240	5.20E-28	2.66E-31	8.14E-30	4.17E-33	1.18E-29	6.03E-33
Pu-240MF	8.51E-03	9.84E-01	7.84E-03	8.80E-01	8.04E-03	9.03E-01
Pu-242	1.64E-28	6.86E-31	2.63E-30	1.10E-32	3.79E-30	1.58E-32
Pu-242MF	1.06E-03	1.00E+00	1.06E-03	9.69E-01	1.06E-03	9.69E-01
Th-232	1.39E-39	1.15E-40	2.14E-41	1.76E-42	3.11E-41	2.55E-42
Th-232MF	5.34E-05	1.00E+00	5.34E-05	1.00E+00	5.34E-05	1.00E+00

[a] The "MF" designation refers to the mobile fraction

Aquifer Concentrations

Groundwater concentrations for the five closure scenarios are presented in Tables 16 through 20. Radioactive progeny were only computed for the actinide immobile fractions, with the exception of Pu-238. Unsaturated transit times of the immobile fractions were short enough such that little progeny generation would occur. However, the half-life of Pu-238 is relatively short and its progeny (U-234) may be important, and was therefore included in the Pu-238 mobile fraction dose. In general, the enhanced cover provides the greatest protection (i.e., lowest groundwater concentrations) while the cover remains intact. The main impact the engineered covers have over

the site soils cover is that they prevent further migration of the relatively mobile radionuclides and thereby, reduce concentrations and dose while the cover is intact. Following cover failure, groundwater concentrations of the less mobile actinides are almost the same for each closure scenario.

Some of the results in Table 16 may seem counterintuitive. For example, the maximum U-238MF concentration of for the enhanced cover is greater than that of the proposed cover, despite the fact that the enhanced cover has a lower infiltration rate. Figure 18 shows the U-238MF groundwater concentrations as a function of time for the two covers. For the enhanced cover, U-238 activity builds up in the unsaturated zone while the cover is intact. When failure occurs, the activity built up in the unsaturated zone releases in a relatively short period of time. For the proposed cover, releases to the aquifer are higher than the enhanced cover while the cover is intact. When the cover fails, there is less activity released to the aquifer compared the enhanced cover, and therefore the *maximum* concentration is less.

Table 16. Groundwater Concentrations for the Site Soils Cover for Closure in 2056

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
H-3		9.0E-05	5.4E-08	5.6E-17	2.0E-39	0.0E+00	9.0E-05
C-14		1.5E-13	1.9E-10	1.3E-08	8.0E-07	1.1E-06	1.1E-06
C-14MF		6.8E-08	2.5E-08	4.5E-12	2.2E-20	7.5E-43	6.8E-08
Cl-36		1.3E-08	1.4E-08	1.1E-08	1.8E-09	4.8E-10	1.4E-08
I-129		1.4E-16	2.4E-13	1.8E-11	1.7E-09	3.2E-09	3.2E-09
I-129MF		1.0E-10	3.5E-11	5.6E-15	2.9E-23	1.6E-45	1.0E-10
Tc-99		2.2E-07	2.2E-07	1.8E-07	3.0E-08	7.9E-09	2.2E-07
U-238		5.7E-25	1.3E-21	3.5E-19	1.5E-15	4.0E-13	3.2E-08
	U-234	4.2E-28	2.3E-24	1.5E-21	3.0E-17	1.6E-14	1.5E-08
	Th-230	6.4E-38	1.5E-33	1.5E-29	1.0E-23	2.1E-20	1.2E-11
	Ra-226	1.4E-35	3.6E-31	3.8E-27	3.3E-21	5.6E-18	6.4E-10
	Pb-210	4.8E-38	1.2E-33	1.3E-29	1.1E-23	1.9E-20	2.2E-12
U-238MF		1.5E-08	4.6E-09	6.8E-13	3.5E-21	1.9E-43	1.5E-08
U-235		5.6E-26	2.5E-22	9.6E-20	5.5E-16	1.5E-13	2.9E-09
	Pa-231	1.2E-30	1.4E-26	3.6E-23	1.5E-18	7.8E-16	3.1E-10
	Ac-227	1.4E-34	2.0E-30	4.9E-27	2.0E-22	9.9E-20	3.9E-14
U-235MF		3.0E-10	9.2E-11	1.4E-14	7.0E-23	3.8E-45	3.0E-10
U-234		5.1E-25	2.3E-21	8.8E-19	5.0E-15	1.3E-12	2.2E-08
	Th-230	7.8E-35	1.6E-30	1.0E-26	2.2E-21	2.2E-18	1.7E-11
	Ra-226	1.8E-32	3.8E-28	2.7E-24	6.9E-19	5.7E-16	8.6E-10
	Pb-210	6.0E-35	1.3E-30	9.3E-27	2.4E-21	1.9E-18	2.9E-12
U-234MF		2.8E-09	8.4E-10	1.3E-13	6.4E-22	3.5E-44	2.8E-09
Th-230		4.0E-66	7.7E-62	3.0E-58	2.5E-53	3.9E-50	1.1E-37
	Ra-226	3.4E-42	8.1E-38	2.3E-34	2.2E-30	4.2E-28	3.5E-25
	Pb-210	1.2E-44	2.7E-40	7.8E-37	7.5E-33	1.4E-30	1.2E-27
Th-230MF		2.6E-13	8.7E-14	1.4E-17	6.9E-26	3.7E-48	2.6E-13
Ra-226		1.7E-37	1.3E-33	1.8E-30	6.5E-27	6.0E-25	1.9E-23

Table 16. Groundwater Concentrations for the Site Soils Cover for Closure in 2056

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
	Pb-210	5.8E-40	4.6E-36	6.1E-33	2.2E-29	2.0E-27	6.4E-26
Ra-226MF		2.8E-11	9.8E-12	1.6E-15	6.8E-24	6.9E-47	2.8E-11
Pu-238		1.6E-51	1.5E-48	6.0E-47	4.7E-47	6.3E-56	6.0E-47
	U-234	9.7E-28	6.0E-24	3.5E-21	3.3E-17	1.2E-14	3.0E-10
	Th-230	1.4E-37	3.8E-33	3.6E-29	1.3E-23	1.8E-20	2.3E-13
	Ra-226	3.2E-35	9.2E-31	9.7E-27	4.2E-21	4.7E-18	1.1E-11
	Pb-210	1.1E-37	3.1E-33	3.3E-29	1.4E-23	1.6E-20	3.9E-14
Pu-238MF		4.9E-10	3.5E-11	5.3E-16	2.5E-26	5.9E-62	4.9E-10
	U-234MF	3.6E-13	8.6E-14	1.1E-17	5.2E-26	3.0E-48	3.6E-13
Pu-239		4.7E-51	3.1E-47	5.7E-44	2.6E-39	2.8E-36	2.9E-25
	U-235	1.5E-31	1.0E-27	7.2E-25	1.0E-20	5.8E-18	8.0E-12
	Pa-231	3.0E-36	5.3E-32	2.4E-28	2.5E-23	2.6E-20	1.1E-12
	Ac-227	3.6E-40	7.8E-36	3.3E-32	3.3E-27	3.4E-24	1.3E-16
Pu-239MF		5.7E-10	1.1E-10	1.1E-14	5.4E-23	2.7E-45	5.7E-10
Pu-240		2.0E-51	1.3E-47	2.3E-44	7.7E-40	5.8E-37	6.5E-29
	U-236	2.0E-30	1.4E-26	9.4E-24	1.3E-19	7.4E-17	4.4E-11
	Th-232	1.5E-45	4.4E-41	5.0E-37	2.6E-31	5.2E-28	2.6E-19
	Ra-228	5.9E-42	7.1E-38	1.4E-34	9.7E-30	9.5E-27	2.5E-19
	Th-228	1.8E-43	2.2E-39	4.2E-36	6.1E-32	1.0E-44	6.1E-32
Pu-240MF		2.4E-10	4.6E-11	4.8E-15	2.2E-23	7.9E-46	2.4E-10
Pu242		2.5E-52	1.6E-48	3.1E-45	1.6E-40	2.0E-37	2.2E-25
	U-238	1.5E-57	1.2E-53	1.1E-50	6.2E-46	9.7E-43	4.1E-30
	U-234	3.0E-38	5.7E-34	8.8E-31	3.2E-26	5.0E-23	1.2E-14
	Th-230	3.6E-48	2.7E-43	7.3E-39	9.7E-33	4.5E-29	4.5E-18
	Ra-226	8.1E-46	6.5E-41	1.9E-36	3.2E-30	1.3E-26	2.7E-16
	Pb-210	2.8E-48	2.2E-43	6.6E-39	1.1E-32	4.3E-29	9.2E-19
Pu-242MF		3.0E-11	5.7E-12	6.2E-16	3.0E-24	1.6E-46	3.0E-11
Th-232		5.0E-65	5.9E-61	2.1E-57	1.7E-52	2.7E-49	1.8E-36
	Ra-228	1.3E-47	1.1E-48	1.3E-48	1.6E-50	8.2E-48	3.3E-36
	Th-228	4.0E-49	3.2E-50	3.8E-50	1.3E-52	8.7E-66	4.0E-49
Th-232MF		1.5E-12	5.2E-13	9.2E-17	4.8E-25	2.6E-47	1.5E-12

a. The "MF" designation refers to mobile fraction

Table 17. Groundwater Concentrations for the Enhanced Cover for Closure in 2003

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
H-3		8.0E-05	4.1E-10	2.3E-16	2.8E-29	0.0E+00	8.0E-05
C-14		2.4E-18	1.8E-18	5.4E-17	3.0E-08	6.0E-07	6.0E-07
C-14MF		4.7E-09	1.3E-09	9.6E-09	1.6E-08	4.3E-28	1.6E-08

Table 17. Groundwater Concentrations for the Enhanced Cover for Closure in 2003

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
Cl-36		1.1E-10	5.6E-11	7.7E-10	3.4E-09	1.2E-09	3.4E-09
I-129		4.0E-22	3.7E-22	1.9E-20	7.4E-11	2.8E-09	3.0E-09
I-129MF		2.8E-12	1.5E-12	1.6E-11	2.7E-11	9.0E-31	2.7E-11
Tc-99		4.3E-09	1.5E-09	1.5E-08	5.2E-08	1.8E-08	5.2E-08
U-238		1.9E-29	1.6E-29	3.7E-28	3.1E-18	1.6E-14	3.1E-08
	U-234	1.0E-32	1.8E-32	1.0E-30	9.9E-20	1.3E-15	1.5E-08
	Th-230	3.5E-41	4.2E-40	1.1E-38	1.0E-26	8.8E-22	1.1E-11
	Ra-226	6.0E-39	9.7E-38	7.1E-36	3.7E-24	2.6E-19	6.1E-10
	Pb-210	2.1E-41	3.3E-40	2.4E-38	1.3E-26	8.8E-22	2.1E-12
U-238MF		9.3E-10	3.1E-10	2.5E-09	3.9E-09	1.1E-28	3.9E-09
U-235		7.6E-31	6.5E-31	1.8E-29	9.6E-19	6.0E-15	2.8E-09
	Pa-231	1.6E-34	5.0E-34	9.4E-33	1.4E-21	2.2E-17	3.1E-10
	Ac-227	2.1E-38	6.2E-38	1.7E-36	1.8E-25	2.8E-21	3.8E-14
U-235MF		1.9E-11	6.3E-12	5.1E-11	8.0E-11	2.2E-30	8.0E-11
U-234		6.9E-30	5.9E-30	1.6E-28	8.6E-18	5.3E-14	2.1E-08
	Th-230	3.6E-38	2.6E-37	2.9E-36	1.1E-24	4.4E-20	1.7E-11
	Ra-226	6.3E-36	5.8E-35	2.1E-33	3.8E-22	1.3E-17	8.4E-10
	Pb-210	2.1E-38	2.0E-37	7.2E-36	1.3E-24	4.3E-20	2.9E-12
U-234MF		1.7E-10	5.7E-11	4.6E-10	7.3E-10	2.0E-29	7.3E-10
Th-230		3.4E-70	1.1E-69	5.5E-69	1.1E-56	5.3E-52	5.1E-38
	Ra-226	3.7E-47	4.5E-47	4.2E-46	8.8E-33	5.1E-29	2.4E-25
	Pb-210	1.3E-49	1.5E-49	1.4E-48	3.0E-35	1.7E-31	8.1E-28
Th-230MF		8.1E-15	4.0E-15	4.0E-14	6.8E-14	2.1E-33	6.8E-14
Ra-226		9.8E-42	1.1E-41	7.6E-41	4.3E-30	9.7E-27	2.7E-24
	Pb-210	3.3E-44	3.7E-44	2.6E-43	1.5E-32	3.3E-29	9.2E-27
Ra-226MF		1.5E-12	4.8E-13	3.0E-12	4.8E-12	3.9E-32	4.8E-12
Pu-238		2.4E-55	2.3E-55	7.9E-56	3.2E-55	3.9E-59	3.2E-55
	U-234	9.1E-33	8.1E-33	2.5E-31	6.3E-20	5.5E-16	2.9E-10
	Th-230	4.7E-41	3.5E-40	4.1E-39	7.0E-27	4.2E-22	2.2E-13
	Ra-226	8.1E-39	7.9E-38	3.0E-36	2.5E-24	1.2E-19	1.1E-11
	Pb-210	2.8E-41	2.7E-40	1.0E-38	8.4E-27	4.2E-22	3.8E-14
Pu-238MF		1.6E-10	7.1E-12	5.8E-13	6.1E-14	3.4E-47	1.6E-10
	U-234MF	3.5E-14	1.7E-14	1.0E-13	1.3E-13	1.7E-33	1.3E-13
Pu-239		6.9E-55	1.8E-54	8.8E-54	1.4E-42	4.5E-38	1.4E-25
	U-235	1.4E-36	1.2E-36	3.8E-35	6.3E-23	9.0E-19	7.9E-12
	Pa-231	3.0E-40	9.5E-40	1.9E-38	7.8E-26	3.0E-21	1.0E-12
	Ac-227	3.9E-44	1.2E-43	3.3E-42	1.1E-29	3.9E-25	1.3E-16
Pu-239MF		1.0E-10	2.2E-11	1.1E-10	1.4E-10	1.5E-30	1.4E-10
Pu-240		2.9E-55	7.4E-55	3.5E-54	4.2E-43	9.0E-39	3.0E-29
	U-236	1.8E-35	1.6E-35	5.0E-34	7.8E-22	1.1E-17	4.3E-11
	Th-232	5.0E-49	3.7E-48	4.4E-47	4.2E-34	4.1E-29	2.5E-19

Table 17. Groundwater Concentrations for the Enhanced Cover for Closure in 2003

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
	Ra-228	1.9E-46	4.4E-46	1.3E-44	3.3E-32	1.1E-27	2.5E-19
	Th-228	5.8E-48	1.3E-47	4.0E-46	5.4E-35	1.1E-45	5.4E-35
Pu-240MF		4.4E-11	9.5E-12	4.4E-11	5.5E-11	4.5E-31	5.5E-11
Pu242		3.7E-56	9.5E-56	4.8E-55	8.6E-44	3.1E-39	1.0E-25
	U-238	2.4E-62	4.3E-62	2.7E-60	6.0E-49	2.0E-44	1.9E-30
	U-234	1.2E-43	1.1E-43	4.0E-42	8.2E-28	1.9E-23	1.1E-14
	Th-230	6.0E-52	4.7E-51	5.8E-50	7.4E-35	1.2E-29	4.2E-18
	Ra-226	1.0E-49	1.1E-48	4.2E-47	2.7E-32	3.5E-27	2.5E-16
	Pb-210	3.5E-52	3.6E-51	1.4E-49	9.1E-35	1.2E-29	8.5E-19
Pu-242MF		5.5E-12	1.2E-12	6.0E-12	7.5E-12	9.2E-32	7.5E-12
Th-232		1.2E-68	3.6E-68	1.6E-67	7.1E-56	3.5E-51	7.7E-37
	Ra-228	9.8E-48	3.1E-56	5.4E-59	4.2E-53	1.9E-49	1.5E-36
	Th-228	3.0E-49	9.3E-58	1.6E-60	2.1E-55	2.0E-67	3.0E-49
Th-232MF		1.1E-13	3.1E-14	2.4E-13	3.9E-13	1.5E-32	3.9E-13

a. The "MF" designation refers to mobile fraction

Table 18. Groundwater Concentrations for the Enhanced Cover for Closure in 2056

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
H-3		8.0E-05	4.1E-10	2.3E-16	3.6E-29	0.0E+00	8.0E-05
C-14		2.4E-18	1.8E-18	5.4E-17	3.5E-08	7.6E-07	7.6E-07
C-14MF		4.7E-09	1.3E-09	9.6E-09	1.6E-08	4.3E-28	1.6E-08
Cl-36		1.1E-10	5.6E-11	7.7E-10	3.5E-09	1.2E-09	3.5E-09
I-129		4.0E-22	3.7E-22	1.9E-20	7.7E-11	2.9E-09	3.2E-09
I-129MF		2.8E-12	1.5E-12	1.6E-11	2.7E-11	9.0E-31	2.7E-11
Tc-99		4.3E-09	1.5E-09	1.5E-08	5.7E-08	2.0E-08	5.7E-08
U-238		1.9E-29	1.6E-29	3.7E-28	3.1E-18	1.6E-14	3.1E-08
	U-234	1.0E-32	1.8E-32	1.0E-30	9.9E-20	1.3E-15	1.5E-08
	Th-230	3.5E-41	4.2E-40	1.1E-38	1.0E-26	8.8E-22	1.1E-11
	Ra-226	6.0E-39	9.7E-38	7.1E-36	3.7E-24	2.6E-19	6.1E-10
	Pb-210	2.1E-41	3.3E-40	2.4E-38	1.3E-26	8.8E-22	2.1E-12
U-238MF		9.3E-10	3.1E-10	2.5E-09	3.9E-09	1.1E-28	3.9E-09
U-235		7.6E-31	6.5E-31	1.8E-29	9.6E-19	6.0E-15	2.8E-09
	Pa-231	1.6E-34	5.0E-34	9.4E-33	1.4E-21	2.2E-17	3.1E-10
	Ac-227	2.1E-38	6.2E-38	1.7E-36	1.8E-25	2.8E-21	3.8E-14
U-235MF		1.9E-11	6.3E-12	5.1E-11	8.0E-11	2.2E-30	8.0E-11
U-234		6.9E-30	5.9E-30	1.6E-28	8.7E-18	5.4E-14	2.2E-08
	Th-230	3.6E-38	2.6E-37	2.9E-36	1.1E-24	4.4E-20	1.7E-11
	Ra-226	6.3E-36	5.8E-35	2.1E-33	3.8E-22	1.3E-17	8.4E-10

Table 18. Groundwater Concentrations for the Enhanced Cover for Closure in 2056

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
	Pb-210	2.1E-38	2.0E-37	7.2E-36	1.3E-24	4.3E-20	2.9E-12
U-234MF		1.7E-10	5.7E-11	4.6E-10	7.3E-10	2.0E-29	7.3E-10
Th-230		3.4E-70	1.1E-69	5.5E-69	1.1E-56	5.3E-52	5.1E-38
	Ra-226	3.7E-47	4.5E-47	4.2E-46	8.8E-33	5.1E-29	2.4E-25
	Pb-210	1.3E-49	1.5E-49	1.4E-48	3.0E-35	1.7E-31	8.1E-28
Th-230MF		8.1E-15	4.0E-15	4.0E-14	6.8E-14	2.1E-33	6.8E-14
Ra-226		9.8E-42	1.1E-41	7.6E-41	5.1E-30	1.2E-26	3.7E-24
	Pb-210	3.3E-44	3.7E-44	2.6E-43	1.7E-32	4.2E-29	1.3E-26
Ra-226MF		1.5E-12	4.8E-13	3.0E-12	4.8E-12	3.9E-32	4.8E-12
Pu-238		2.4E-55	2.3E-55	7.9E-56	3.2E-55	3.9E-59	3.2E-55
	U-234	9.1E-33	8.1E-33	2.5E-31	6.3E-20	5.5E-16	2.9E-10
	Th-230	4.7E-41	3.5E-40	4.1E-39	7.0E-27	4.2E-22	2.2E-13
	Ra-226	8.1E-39	7.9E-38	3.0E-36	2.5E-24	1.2E-19	1.1E-11
	Pb-210	2.8E-41	2.7E-40	1.0E-38	8.4E-27	4.2E-22	3.8E-14
Pu-238MF		1.6E-10	7.1E-12	5.8E-13	6.1E-14	3.4E-47	1.6E-10
	U-234MF	3.5E-14	1.7E-14	1.0E-13	1.3E-13	1.7E-33	1.3E-13
Pu-239		6.9E-55	1.8E-54	8.8E-54	1.4E-42	4.5E-38	1.4E-25
	U-235	1.4E-36	1.2E-36	3.8E-35	6.3E-23	9.0E-19	7.9E-12
	Pa-231	3.0E-40	9.5E-40	1.9E-38	7.8E-26	3.0E-21	1.0E-12
	Ac-227	3.9E-44	1.2E-43	3.3E-42	1.1E-29	3.9E-25	1.3E-16
Pu-239MF		1.0E-10	2.2E-11	1.1E-10	1.4E-10	1.5E-30	1.4E-10
Pu-240		2.9E-55	7.4E-55	3.5E-54	4.2E-43	9.0E-39	3.0E-29
	U-236	1.8E-35	1.6E-35	5.0E-34	7.8E-22	1.1E-17	4.3E-11
	Th-232	5.0E-49	3.7E-48	4.4E-47	4.2E-34	4.1E-29	2.5E-19
	Ra-228	1.9E-46	4.4E-46	1.3E-44	3.3E-32	1.1E-27	2.5E-19
	Th-228	5.8E-48	1.3E-47	4.0E-46	5.4E-35	1.1E-45	5.4E-35
Pu-240MF		4.4E-11	9.5E-12	4.4E-11	5.5E-11	4.5E-31	5.5E-11
Pu242		3.7E-56	9.5E-56	4.8E-55	8.6E-44	3.1E-39	1.0E-25
	U-238	2.4E-62	4.3E-62	2.7E-60	6.0E-49	2.0E-44	1.9E-30
	U-234	1.2E-43	1.1E-43	4.0E-42	8.2E-28	1.9E-23	1.1E-14
	Th-230	6.0E-52	4.7E-51	5.8E-50	7.5E-35	1.2E-29	4.2E-18
	Ra-226	1.0E-49	1.1E-48	4.2E-47	2.7E-32	3.5E-27	2.5E-16
	Pb-210	3.5E-52	3.6E-51	1.4E-49	9.1E-35	1.2E-29	8.5E-19
Pu-242MF		5.5E-12	1.2E-12	6.0E-12	7.5E-12	9.2E-32	7.5E-12
Th-232		1.2E-68	3.6E-68	1.6E-67	7.3E-56	3.7E-51	8.0E-37
	Ra-228	9.8E-48	3.1E-56	5.5E-59	4.3E-53	2.0E-49	1.5E-36
	Th-228	3.0E-49	9.3E-58	1.7E-60	2.1E-55	2.1E-67	3.0E-49
Th-232MF		1.1E-13	3.1E-14	2.4E-13	3.9E-13	1.5E-32	3.9E-13

a. The "MF" designation refers to mobile fraction

Table 19. Groundwater Concentrations for the Enhanced Cover for Closure in 2215

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
H-3		8.0E-05	4.1E-10	2.3E-16	4.0E-26	0.0E+00	8.0E-05
C-14		2.4E-18	1.8E-18	5.4E-17	5.1E-08	1.2E-06	1.2E-06
C-14MF		4.7E-09	1.3E-09	9.6E-09	1.6E-08	4.3E-28	1.6E-08
Cl-36		1.1E-10	5.6E-11	7.7E-10	3.8E-09	1.3E-09	3.8E-09
I-129		4.0E-22	3.7E-22	1.9E-20	8.5E-11	3.4E-09	3.7E-09
I-129MF		2.8E-12	1.5E-12	1.6E-11	2.7E-11	9.0E-31	2.7E-11
Tc-99		4.3E-09	1.5E-09	1.5E-08	7.2E-08	2.5E-08	7.2E-08
U-238		1.9E-29	1.6E-29	3.7E-28	3.1E-18	1.6E-14	3.1E-08
	U-234	1.0E-32	1.8E-32	1.0E-30	1.0E-19	1.3E-15	1.6E-08
	Th-230	3.5E-41	4.2E-40	1.1E-38	1.0E-26	8.8E-22	1.1E-11
	Ra-226	6.0E-39	9.7E-38	7.1E-36	3.7E-24	2.6E-19	6.2E-10
	Pb-210	2.1E-41	3.3E-40	2.4E-38	1.3E-26	8.8E-22	2.1E-12
U-238MF		9.3E-10	3.1E-10	2.5E-09	3.9E-09	1.1E-28	3.9E-09
U-235		7.6E-31	6.5E-31	1.8E-29	9.7E-19	6.1E-15	2.8E-09
	Pa-231	1.6E-34	5.0E-34	9.4E-33	1.4E-21	2.2E-17	3.1E-10
	Ac-227	2.1E-38	6.2E-38	1.7E-36	1.8E-25	2.9E-21	3.9E-14
U-235MF		1.9E-11	6.3E-12	5.1E-11	8.0E-11	2.2E-30	8.0E-11
U-234		6.9E-30	5.9E-30	1.6E-28	8.7E-18	5.4E-14	2.2E-08
	Th-230	3.6E-38	2.6E-37	2.9E-36	1.1E-24	4.4E-20	1.7E-11
	Ra-226	6.3E-36	5.8E-35	2.1E-33	3.8E-22	1.3E-17	8.5E-10
	Pb-210	2.1E-38	2.0E-37	7.2E-36	1.3E-24	4.3E-20	2.9E-12
U-234MF		1.7E-10	5.7E-11	4.6E-10	7.3E-10	2.0E-29	7.3E-10
Th-230		3.4E-70	1.1E-69	5.5E-69	1.1E-56	5.3E-52	5.1E-38
	Ra-226	3.7E-47	4.5E-47	4.2E-46	8.8E-33	5.1E-29	2.4E-25
	Pb-210	1.3E-49	1.5E-49	1.4E-48	3.0E-35	1.7E-31	8.1E-28
Th-230MF		8.1E-15	4.0E-15	4.0E-14	6.8E-14	2.1E-33	6.8E-14
Ra-226		9.8E-42	1.1E-41	7.6E-41	7.5E-30	2.1E-26	6.6E-24
	Pb-210	3.3E-44	3.7E-44	2.6E-43	2.6E-32	7.1E-29	2.3E-26
Ra-226MF		1.5E-12	4.8E-13	3.0E-12	4.8E-12	3.9E-32	4.8E-12
Pu-238		2.4E-55	2.3E-55	7.9E-56	3.2E-55	3.9E-59	3.2E-55
	U-234	9.1E-33	8.1E-33	2.5E-31	6.4E-20	5.5E-16	2.9E-10
	Th-230	4.7E-41	3.5E-40	4.1E-39	7.0E-27	4.3E-22	2.2E-13
	Ra-226	8.1E-39	7.9E-38	3.0E-36	2.5E-24	1.2E-19	1.1E-11
	Pb-210	2.8E-41	2.7E-40	1.0E-38	8.4E-27	4.2E-22	3.8E-14
Pu-238MF		1.6E-10	7.1E-12	5.8E-13	6.1E-14	3.4E-47	1.6E-10
	U-234MF	3.5E-14	1.7E-14	1.0E-13	1.3E-13	1.7E-33	1.3E-13
Pu-239		6.9E-55	1.8E-54	8.8E-54	1.4E-42	4.5E-38	1.4E-25
	U-235	1.4E-36	1.2E-36	3.8E-35	6.3E-23	9.0E-19	8.0E-12
	Pa-231	3.0E-40	9.5E-40	1.9E-38	7.9E-26	3.0E-21	1.0E-12
	Ac-227	3.9E-44	1.2E-43	3.3E-42	1.1E-29	3.9E-25	1.3E-16

Table 19. Groundwater Concentrations for the Enhanced Cover for Closure in 2215

Radionuclide[a]	Progeny	Maximum 0-250 yr (Ci m^{-3})	Maximum 250-500 yr (Ci m^{-3})	Maximum 500-1000 yr (Ci m^{-3})	Maximum 1000-5000 yr (Ci m^{-3})	Maximum 5000-10,000 yr (Ci m^{-3})	Maximum 0-200,000 yr (Ci m^{-3})
Pu-239MF		1.0E-10	2.2E-11	1.1E-10	1.4E-10	1.5E-30	1.4E-10
Pu-240		2.9E-55	7.4E-55	3.5E-54	4.2E-43	9.0E-39	3.0E-29
	U-236	1.8E-35	1.6E-35	5.0E-34	7.8E-22	1.1E-17	4.3E-11
	Th-232	5.0E-49	3.7E-48	4.4E-47	4.2E-34	4.1E-29	2.5E-19
	Ra-228	1.9E-46	4.4E-46	1.3E-44	3.3E-32	1.1E-27	2.5E-19
	Th-228	5.8E-48	1.3E-47	4.0E-46	5.4E-35	1.1E-45	5.4E-35
Pu-240MF		4.4E-11	9.5E-12	4.4E-11	5.5E-11	4.5E-31	5.5E-11
Pu242		3.7E-56	9.5E-56	4.8E-55	8.6E-44	3.1E-39	1.0E-25
	U-238	2.4E-62	4.3E-62	2.7E-60	6.0E-49	2.0E-44	1.9E-30
	U-234	1.2E-43	1.1E-43	4.0E-42	8.2E-28	1.9E-23	1.1E-14
	Th-230	6.0E-52	4.7E-51	5.8E-50	7.5E-35	1.2E-29	4.2E-18
	Ra-226	1.0E-49	1.1E-48	4.2E-47	2.7E-32	3.5E-27	2.5E-16
	Pb-210	3.5E-52	3.6E-51	1.4E-49	9.1E-35	1.2E-29	8.6E-19
Pu-242MF		5.5E-12	1.2E-12	6.0E-12	7.5E-12	9.2E-32	7.5E-12
Th-232		1.2E-68	3.6E-68	1.6E-67	7.6E-56	4.0E-51	9.1E-37
	Ra-228	9.8E-48	3.1E-56	5.8E-59	4.7E-53	2.2E-49	1.7E-36
	Th-228	3.0E-49	9.3E-58	1.8E-60	2.3E-55	2.3E-67	3.0E-49
Th-232MF		1.1E-13	3.1E-14	2.4E-13	3.9E-13	1.5E-32	3.9E-13

a. The "MF" designation refers to mobile fraction

Table 20. Groundwater Concentrations for the Proposed Cover for Closure in 2056

Radionuclide[a]	Progeny	Maximum 0-250 yr (Ci m^{-3})	Maximum 250-500 yr (Ci m^{-3})	Maximum 500-1000 yr (Ci m^{-3})	Maximum 1000-5000 yr (Ci m^{-3})	Maximum 5000-10,000 yr (Ci m^{-3})	Maximum 0-200,000 yr (Ci m^{-3})
H-3		8.0E-05	7.9E-10	1.4E-15	6.6E-29	0.0E+00	8.0E-05
C-14		3.6E-18	1.6E-17	1.3E-15	5.2E-08	7.8E-07	7.8E-07
C-14MF		4.7E-09	4.3E-09	1.3E-08	1.1E-08	1.9E-29	1.3E-08
Cl-36		1.1E-10	2.8E-10	2.3E-09	3.5E-09	1.1E-09	3.5E-09
I-129		7.6E-22	4.6E-21	7.6E-19	1.1E-10	3.0E-09	3.2E-09
I-129MF		2.9E-12	6.3E-12	2.3E-11	1.9E-11	4.0E-32	2.3E-11
Tc-99		4.3E-09	5.9E-09	3.9E-08	5.7E-08	1.8E-08	5.7E-08
U-238		2.8E-29	1.2E-28	8.2E-27	5.8E-18	2.2E-14	3.2E-08
	U-234	1.8E-32	1.6E-31	2.3E-29	1.6E-19	1.5E-15	1.5E-08
	Th-230	4.4E-41	1.3E-39	1.6E-37	1.9E-26	1.1E-21	1.1E-11
	Ra-226	8.4E-39	3.9E-37	6.5E-35	6.5E-24	3.2E-19	6.2E-10
	Pb-210	2.9E-41	1.3E-39	2.2E-37	2.2E-26	1.1E-21	2.1E-12
U-238MF		9.4E-10	1.1E-09	3.1E-09	2.4E-09	4.9E-30	3.1E-09
U-235		1.2E-30	5.6E-30	5.4E-28	1.8E-18	8.1E-15	2.8E-09
	Pa-231	2.2E-34	2.3E-33	1.5E-31	2.7E-21	3.1E-17	3.1E-10
	Ac-227	3.1E-38	3.3E-37	2.4E-35	3.6E-25	3.9E-21	3.8E-14

Table 20. Groundwater Concentrations for the Proposed Cover for Closure in 2056

Radionuclide[a]	Progeny	Maximum 0-250 yr ($Ci\ m^{-3}$)	Maximum 250-500 yr ($Ci\ m^{-3}$)	Maximum 500-1000 yr ($Ci\ m^{-3}$)	Maximum 1000-5000 yr ($Ci\ m^{-3}$)	Maximum 5000-10,000 yr ($Ci\ m^{-3}$)	Maximum 0-200,000 yr ($Ci\ m^{-3}$)
U-235MF		1.9E-11	2.2E-11	6.2E-11	4.9E-11	9.8E-32	6.2E-11
U-234		1.1E-29	5.1E-29	4.9E-27	1.6E-17	7.2E-14	2.2E-08
	Th-230	4.5E-38	7.1E-37	4.1E-35	2.3E-24	6.2E-20	1.7E-11
	Ra-226	8.6E-36	2.2E-34	1.8E-32	8.0E-22	1.8E-17	8.4E-10
	Pb-210	2.9E-38	7.4E-37	6.0E-35	2.7E-24	6.1E-20	2.9E-12
U-234MF		1.7E-10	2.0E-10	5.7E-10	4.5E-10	8.9E-31	5.7E-10
Th-230		4.1E-70	3.0E-69	2.0E-67	2.2E-56	7.7E-52	5.5E-38
	Ra-226	4.9E-47	2.5E-46	2.7E-44	1.3E-32	5.8E-29	2.5E-25
	Pb-210	1.7E-49	8.7E-49	9.2E-47	4.3E-35	2.0E-31	8.4E-28
Th-230MF		8.3E-15	1.6E-14	5.6E-14	4.6E-14	9.4E-35	5.6E-14
Ra-226		1.3E-41	5.1E-41	2.8E-39	1.0E-29	1.8E-26	4.2E-24
	Pb-210	4.3E-44	1.7E-43	9.6E-42	3.5E-32	6.0E-29	1.4E-26
Ra-226MF		1.5E-12	1.6E-12	4.3E-12	3.5E-12	1.8E-33	4.3E-12
Pu-238		2.9E-55	2.9E-55	2.3E-55	2.0E-54	8.0E-59	2.0E-54
	U-234	1.5E-32	7.5E-32	8.4E-30	1.2E-19	7.3E-16	2.9E-10
	Th-230	5.9E-41	9.9E-40	6.4E-38	1.5E-26	5.9E-22	2.2E-13
	Ra-226	1.1E-38	3.0E-37	2.7E-35	5.1E-24	1.7E-19	1.1E-11
	Pb-210	3.8E-41	1.0E-39	9.4E-38	1.7E-26	5.8E-22	3.8E-14
Pu-238MF		1.6E-10	1.2E-11	2.5E-12	2.7E-14	1.5E-48	1.6E-10
	U-234MF	3.5E-14	4.9E-14	8.7E-14	5.7E-14	7.7E-35	8.7E-14
Pu-239		8.5E-55	4.9E-54	2.2E-52	3.0E-42	6.5E-38	1.5E-25
	U-235	2.2E-36	1.2E-35	1.3E-33	9.0E-23	1.0E-18	8.0E-12
	Pa-231	4.1E-40	4.5E-39	3.5E-37	1.2E-25	3.5E-21	1.0E-12
	Ac-227	5.8E-44	6.6E-43	5.4E-41	1.6E-29	4.5E-25	1.3E-16
Pu-239MF		1.0E-10	5.5E-11	9.3E-11	6.0E-11	6.8E-32	1.0E-10
Pu-240		3.6E-55	2.0E-54	9.0E-53	8.8E-43	1.3E-38	3.2E-29
	U-236	2.9E-35	1.5E-34	1.7E-32	1.1E-21	1.2E-17	4.4E-11
	Th-232	6.2E-49	1.1E-47	6.9E-46	6.6E-34	4.8E-29	2.5E-19
	Ra-228	3.2E-46	2.8E-45	2.4E-43	4.9E-32	1.2E-27	2.5E-19
	Th-228	9.8E-48	8.5E-47	7.2E-45	9.2E-35	1.3E-45	9.2E-35
Pu-240MF		4.5E-11	2.3E-11	3.8E-11	2.4E-11	2.0E-32	4.5E-11
Pu242		4.6E-56	2.6E-55	1.2E-53	1.8E-43	4.5E-39	1.1E-25
	U-238	4.2E-62	4.2E-61	7.3E-59	1.2E-48	2.8E-44	2.1E-30
	U-234	2.0E-43	1.2E-42	1.8E-40	1.0E-27	2.1E-23	1.1E-14
	Th-230	7.5E-52	1.4E-50	1.1E-48	9.6E-35	1.3E-29	4.2E-18
	Ra-226	1.4E-49	4.2E-48	4.6E-46	3.4E-32	3.8E-27	2.5E-16
	Pb-210	4.9E-52	1.4E-50	1.6E-48	1.2E-34	1.3E-29	8.6E-19
Pu-242MF		5.5E-12	2.9E-12	5.1E-12	3.3E-12	4.1E-33	5.5E-12
Th-232		1.4E-68	8.9E-68	3.9E-66	1.5E-55	5.4E-51	8.6E-37
	Ra-228	9.9E-48	4.3E-56	5.7E-58	7.4E-53	2.7E-49	1.6E-36
	Th-228	3.0E-49	1.3E-57	1.7E-59	3.8E-55	2.9E-67	3.0E-49

Table 20. Groundwater Concentrations for the Proposed Cover for Closure in 2056

Radionuclide[a]	Progeny	Maximum 0-250 yr (Ci m^{-3})	Maximum 250-500 yr (Ci m^{-3})	Maximum 500-1000 yr (Ci m^{-3})	Maximum 1000-5000 yr (Ci m^{-3})	Maximum 5000-10,000 yr (Ci m^{-3})	Maximum 0-200,000 yr (Ci m^{-3})
Th-232MF		1.1E-13	1.0E-13	3.2E-13	2.7E-13	6.8E-34	3.2E-13

a. The "MF" designation refers to mobile fraction

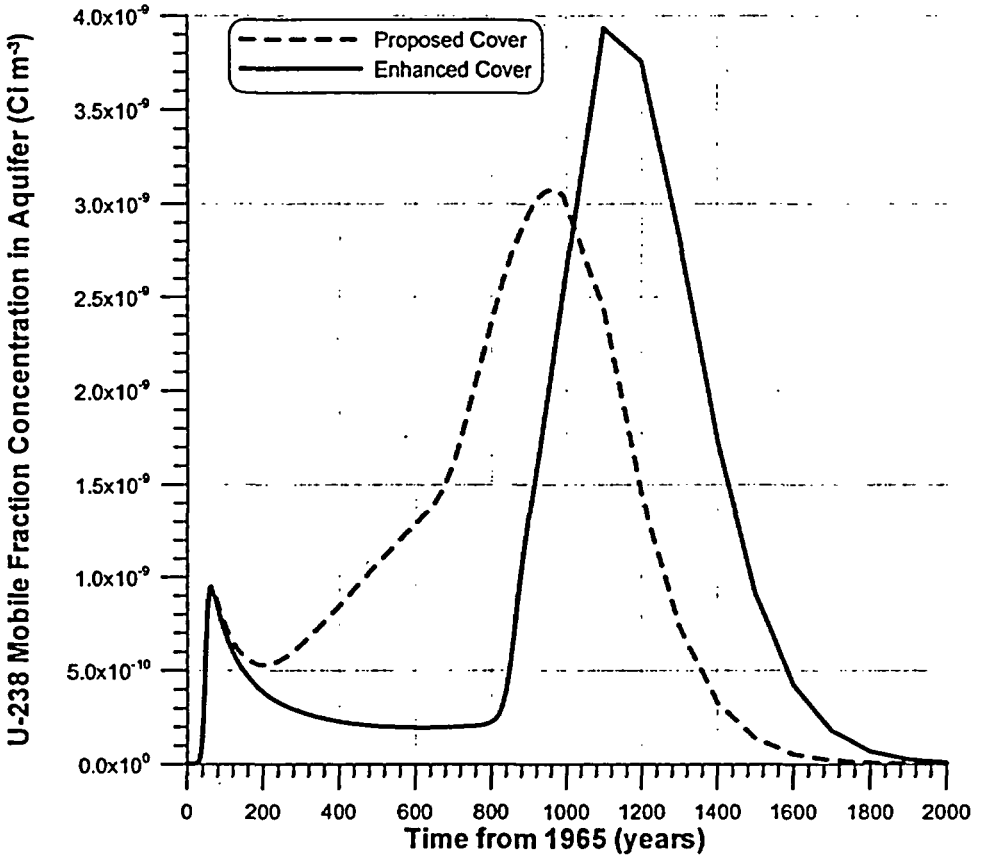

Figure 18. Graph showing U-238 mobile fraction aquifer concentrations for the enhanced and proposed covers. Concentrations while the cover remains intact are lower for the enhanced cover but are higher after cover failure. The area under the two curves is the same.

Drinking water doses as a function of time are presented in Figures 19 through 23. Doses from actinides include doses from all radioactive progeny that form during transport. Dominant dose contributors during specific time periods can be summarized as follows:

- For the 0–100-year time frame, H-3 is the major dose contributor.
- For the 100–1,000 year time frame, U-238 and Tc-99 are the major dose contributors.

- For the 1,000–10,000 year time frame, I-129, C-14, and U-238 are the major dose contributors.

Immobile fraction plutonium and thorium isotopes do not reach their maximum concentration in groundwater until well after 10,000 years.

Total dose as a function of time is presented in Figure 24 for each of the five closure scenarios. This figure illustrates the relative merits of each of the closure options. The site soils cover provides the least amount of protection however, doses are still predicted to be less than 5 mrem yr^{-1} while the cover is intact. Estimated doses while the enhanced and proposed cover are intact are generally less than 1 mrem yr^{-1}. After cover failure, doses from the immobile actinide fraction, C-14, and I-129 are about the same for all closure options. Maximum doses in the 1000–10,000-year time frame are less than 3-mrem yr^{-1} and are driven mainly by U-238, C-14, and I-129.

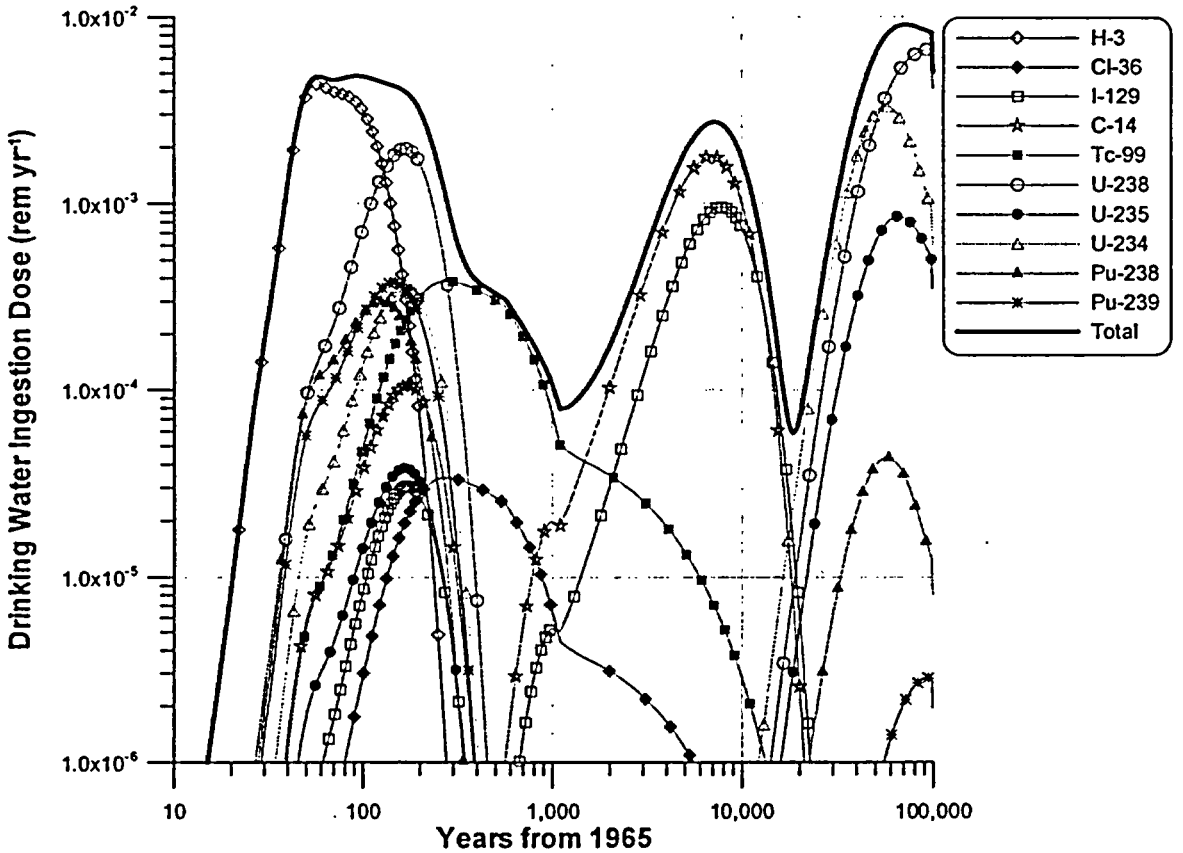

Figure 19. Groundwater ingestion dose as a function of time for the site soils cover for closure in 2056.

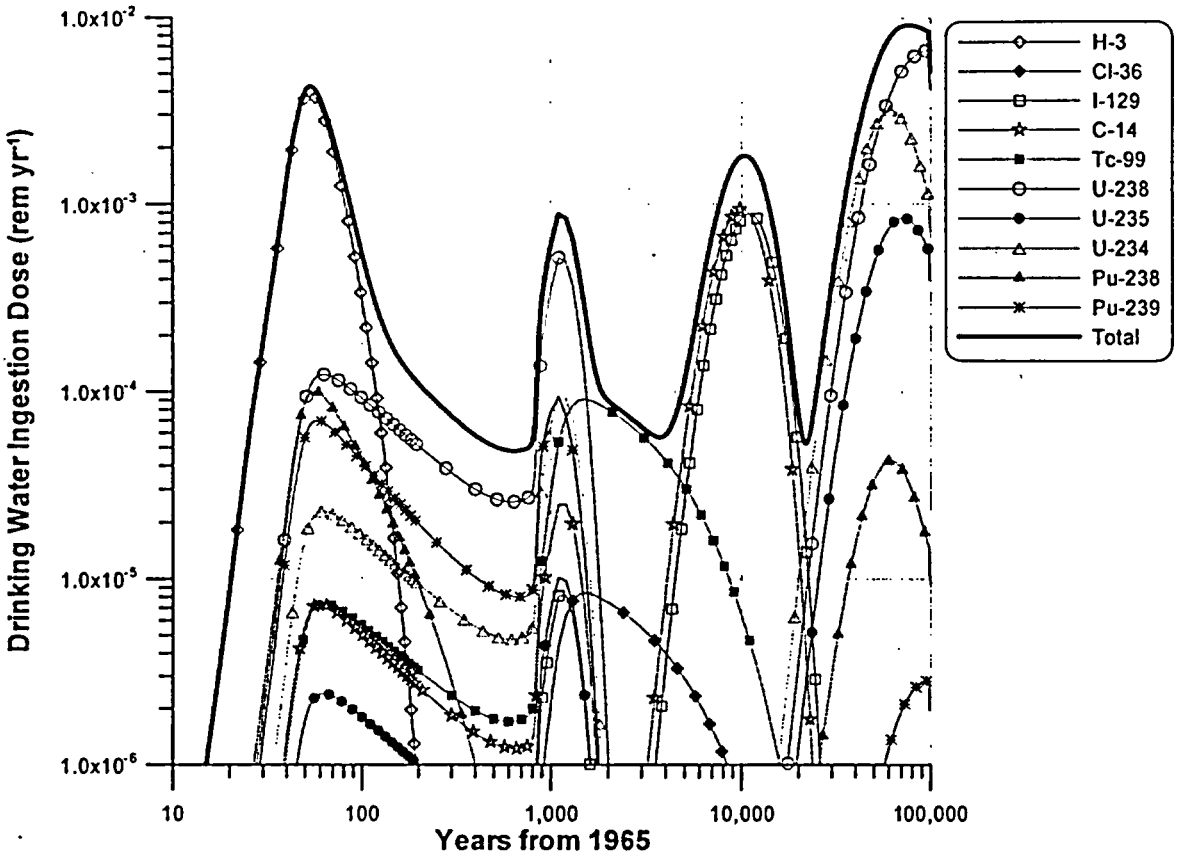

Figure 20. Groundwater ingestion doses as a function of time for the enhanced cover for closure in 2003.

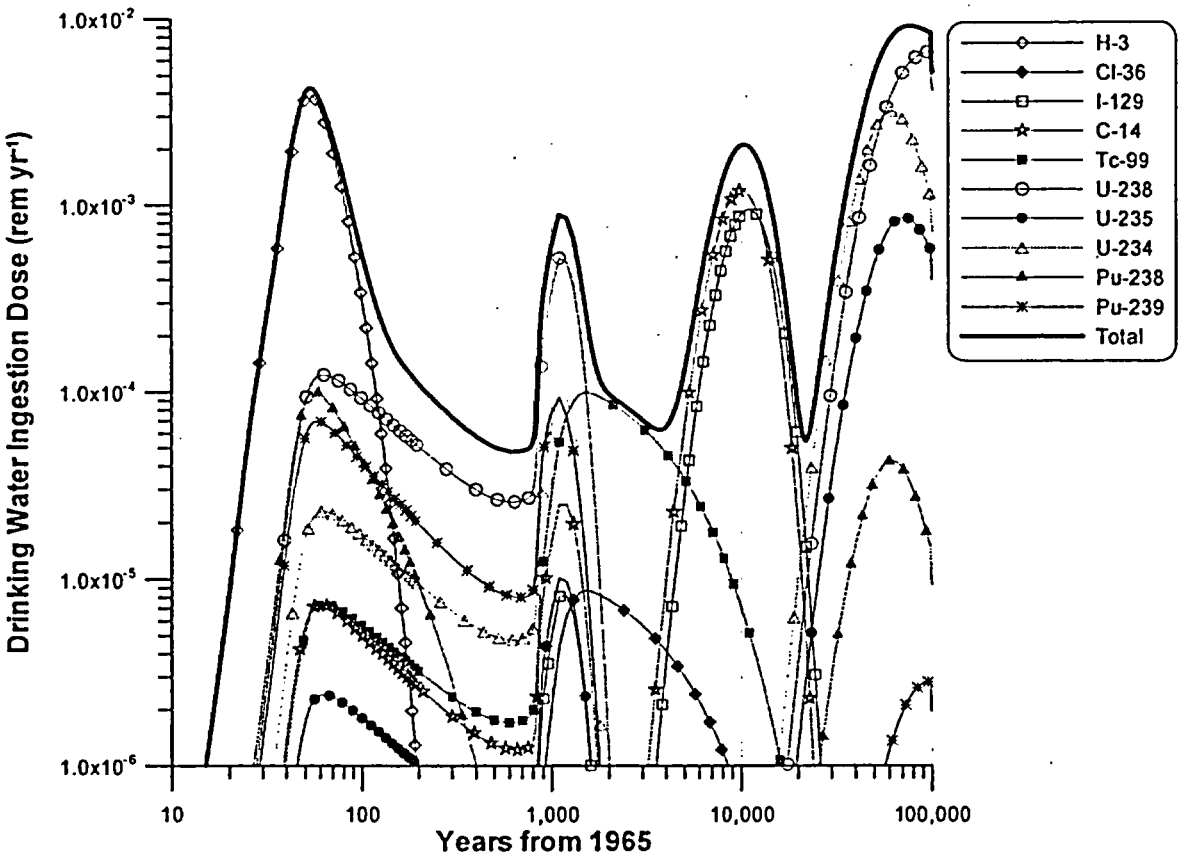

Figure 21. Groundwater ingestion doses as a function of time for the enhanced cover for closure in 2056.

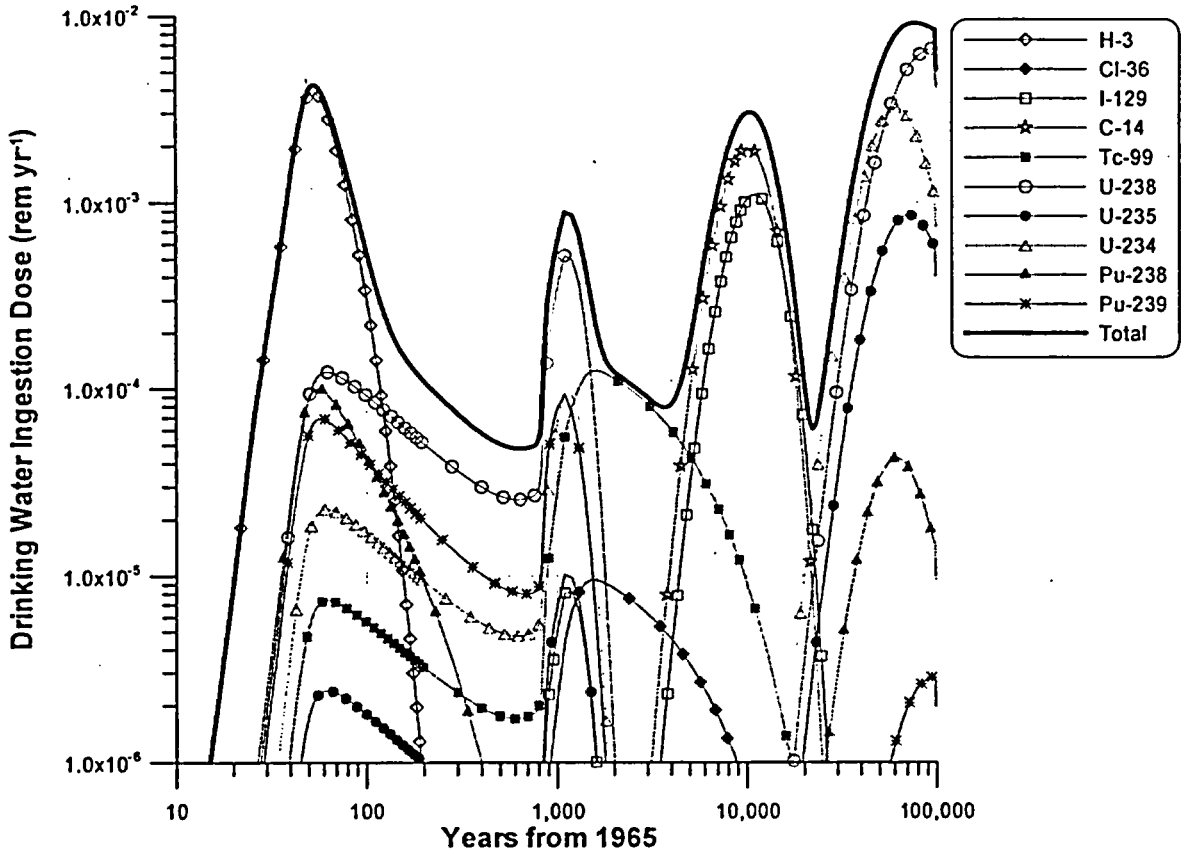

Figure 22. Groundwater ingestion doses as a function of time for the enhanced cover for closure in 2215.

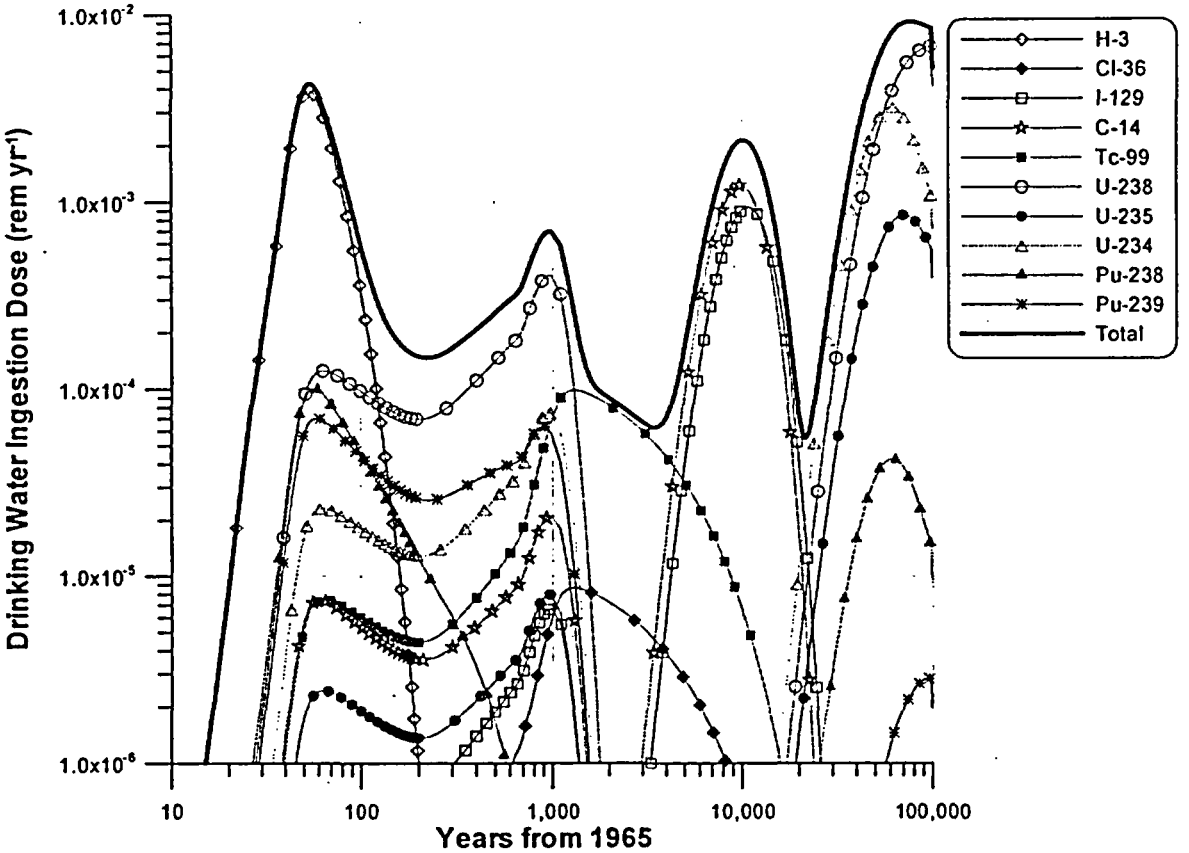

Figure 23. Groundwater ingestion dose as a function of time for the US Ecology proposed cover for closure in 2056.

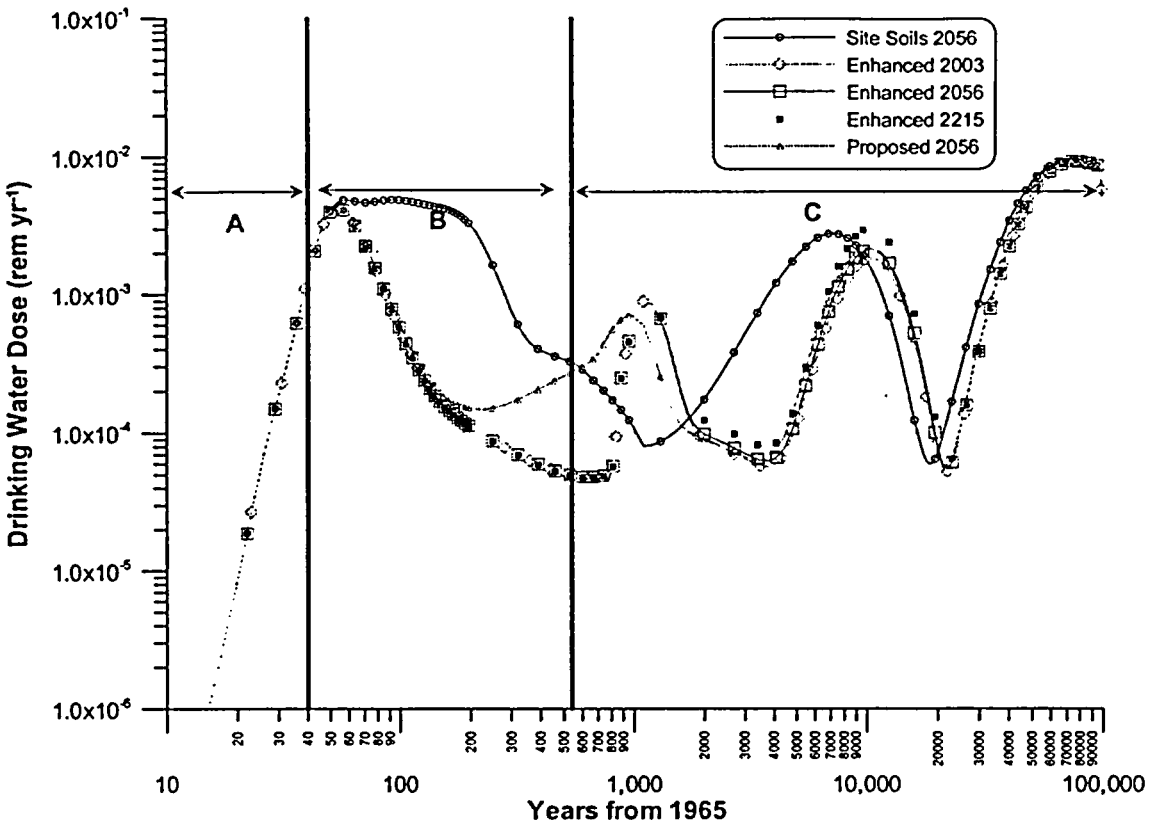

Figure 24. Total drinking water dose as a function of time for the five closure options. Three time periods of interest are shown: A = pre-cover period (year 1965–2005); B = cover period (year 2005–2505); and post cover period (year 2505–100,000).

Comparison With Original DEIS Results

Groundwater concentrations for the original DEIS (Dunkelman 2000) were reported for five of the 15 radionuclides analyzed in this assessment; Cl-36, I-129, Tc-99, U-235, and U-238. There were major differences in the conceptual model for fate and transport, radionuclide inventories, partition coefficients, and assumptions about cover lifetime. Maximum groundwater concentrations in this assessment were in some cases higher and in others, lower than those in the original DEIS (Table 21) for several reasons.

First, the waste disposal history was accounted for in this assessment but was not accounted for in the original DEIS analysis. During active waste disposal in this assessment, infiltration was enhanced due to the presence of open trenches and disturbed soil, resulting in migration of radionuclides from the trenches and into the unsaturated zone before placement of the cover. In the original DEIS, no radionuclide migration was assumed before placement of the cover.

Second, the cover was assumed to only last 500 years (and degrade to natural infiltration over the next 500 years) in this assessment compared to an infinite cover lifetime in the original DEIS. The results of this current assessment show that choice cover design has little impact on the maximum concentration during the 0-10,000 year time of compliance. However, cover design makes a large impact on groundwater concentrations during the time the cover remains intact.

Table 21. Maximum 0–10,000 year Concentrations from the Original DEIS and those from this Assessment for Closure in 2056

			Original DEIS results[a]			Results of this assessment		
Radionuclide	Original DEIS Inventory (Ci)[b]	Inventory used in this assessment (Ci)	Enhanced Cover (pCi L^{-1})	Proposed Cover (pCi L^{-1})	Site Soils Cover (pCi L^{-1})	Enhanced Cover (pCi L^{-1})	Proposed Cover (pCi L^{-1})	Site Soils Cover (pCi L^{-1})
Cl-36	4.910E+00	3.229E+00	2.0E+01	3.6E+01	4.6E+01	3.5E+00	3.5E+00	1.4E+01
I-129	6.010E+00	5.977E+00	1.9E+00	3.9E+00	4.9E+00	2.9E+00	3.2E+00	3.2E+00
Tc-99	6.710E+01	5.510E+01	2.7E+02	4.9E+02	6.3E+02	5.7E+01	5.7E+01	2.2E+02
U-235	1.467E+04	3.058E+01	5.7E-02	2.3E-01	2.3E+00	8.0E-02	6.2E-02	3.0E-01
U-238	2.227E+04	1.510E+03	8.9E-03	3.6E-02	3.6E-01	3.9E+00	3.1E+00	1.5E+01

a. From Section 3.0 of the groundwater pathway analysis for the DEIS (Dunkelman 2000).
b. From Table 8 of the groundwater pathway analysis for the DEIS (Dunkelman 2000).

Third, the cover in this assessment affects infiltration throughout the unsaturated zone whereas, in the original DEIS, it only restricted infiltration through the waste. That is, water fluxes in the unsaturated zone below the waste in the original DEIS were assumed to be the same as natural recharge.

Fourth, in the case of Cl-36 and Tc-99, a calibrated partition coefficient of 0.75 mg L^{-1} was used in the waste whereas in the original DEIS, a partition coefficient of zero was assumed for the waste. This difference is thought to be the major reason why the Cl-36 and Tc-99 concentrations in this assessment are about a factor of 6 lower than those in the original DEIS.

Finally, the maximum uranium isotope concentrations in this assessment during the 0-10,000 year time frame are driven by the mobile fraction of uranium; the immobile fraction peaks after 10,000 years. A mobile fraction was not included in the original DEIS calculations. If we ignore the mobile fraction, uranium isotope concentrations calculated in this assessment during the 0 to 10,000 year time frame are substantially less than those in the original DEIS because disposal inventories were substantially smaller in this assessment compared to those in the original DEIS (500 times smaller for U-235 and 15 times smaller for U-238).

Several other points should be made concerning the uranium isotopes. In the original DEIS, a uranium solubility of 1 mg L^{-1} was used whereas in this assessment, a solubility of 25 mg L^{-1} was used (which was the median estimated value used in the stochastic simulation in Rood 2000). Uranium solubility only affects the U-238 concentration because solubility limited releases are mass (not activity) limited. Also, partition coefficients used in this assessment were corrected for the percent gravel content whereas in the original DEIS, gravel content was not correct for. Gravel-content corrected partition coefficients were about a factor of 2 less than their nominal values.

The transport model used in the original DEIS should not be faulted as being "non-conservative". The difference between the results of this analysis and those in the original DEIS are driven mainly by the assumption of cover lifetime and inclusion of a fraction of the radionuclide inventory that was mobile, and not model formulation. If cover lifetime were assumed to be infinite as was assumed in the original DEIS analysis, then radionuclide concentrations of immobile radionuclides would be lower than the concentrations reported in Tables 16–20.

UNCERTAINTY ANALYSIS

This section presents the methodology and results of the uncertainty and sensitivity analysis performed for the US Ecology Site. An uncertainty analysis evaluates the precision and accuracy of the model. Sensitivity analysis evaluates the sensitivity of model output to variability in model input. Uncertainty in models arises because a) errors in model formulation and b) errors (or uncertainty) in model input parameters (parametric uncertainty). Model formulation errors are inherent in mathematical modeling because environmental models are only simplified representations of complex environmental systems. Errors in model parameterization occur because lack of knowledge about a parameter's true, but unknown value. Ideally, site-specific parameter values should be derived and used in the simulation. In practice, parameter values are often inferred from limited measured data or derived from the literature. Additionally, model parameters may represent time and space scales that differ greatly from what can be measured in the field or laboratory. Natural variability also contributes to parameter uncertainty.

Uncertainty in model formulation can only be evaluated through model validation. Model validation answers the question "Does the model accurately simulate the behavior of the system?". To demonstrate a model is valid, an independent data set is required. Often times, adequate independent data sets are not available and the analyst resorts to model calibration. In model calibration, parameter values are adjusted (within reason) so that model predictions match the field observations as close as possible.

Because this assessment addresses impacts that occur far into the future, it is impossible to validate the model application for future predictions because measurements are unavailable (much in the same way Einstein's theory of the speed of gravity was only recently validated because we lacked the means to measure the necessary quantities). Therefore, model uncertainty is only qualitatively addressed through the calibration procedure discussed earlier in this report. Model calibration only provides a measure of what the model can accurately simulate in the environment for the current time frame. The use of the model for forecasting the release and transport of radionuclides far into the future can never really be truly validated.

A parametric uncertainty analysis quantifies the uncertainty in model output resulting from uncertainty in the model parameters. It is a measure of the precision of the model and cannot address the overall accuracy of the predictions. Parametric uncertainty was evaluated using Monte Carlo simulation combined with simple random sampling techniques. Uncertainty is expressed in terms of a probability density function of the output variable. Information provided by the uncertainty analysis was also used to do the sensitivity analysis. Model sensitivity was evaluated by calculating the rank correlation between the distribution of the output variable and each of the distributions of the input parameters.

Parametric uncertainty analysis uses an estimated frequency distribution of values for each model parameter considered to be uncertain and produces a frequency distribution of model predictions. In Monte Carlo simulation, parameter values are randomly sampled from distributions developed by the analyst. The model is then run and the output variable stored. The process is repeated for multiple model realizations (typically greater than 100) resulting in an

empirical distribution of the output variable. A Perl[3] script was used as the Monte Carlo driver for the simulation and performed the following functions for each Monte Carlo trial:

- sampled parameter values from assigned distributions
- wrote FOLAT and GWSCREEN input files for each of the radionuclides
- executed FOLAT and GWSCREEN models for each radionuclide
- extracted and stored concentrations and doses at specific times from the GWSCREEN output.

The number of radionuclides evaluated in the uncertainty analysis was limited to the primary dose contributors of the deterministic simulation. These radionuclides were H-3, Tc-99, I-129, C-14, U-238, U-238MF, and Pu-239MF. Radioactive progeny from the decay of the immobile U-238 fraction were included in the analysis. The enhanced cover for closure in the year 2056 was the only cover/closure scenario evaluated.

One of the major limitations of this uncertainty analysis is that it did not consider uncertainty in radionuclide inventories or mobile release fractions. Analysis of the mobile release fractions would require an estimate of uncertainty in the source term. Derivation of the uncertainty in the source term was beyond the scope of this project. However, the Perl script written for the uncertainty analysis is certainly amenable to inclusion of this uncertainty in the future. Uncertainty was also not evaluated for the exposure scenario (drinking water ingestion rate) or dose conversion factors.

Infiltration from 1965 to 2005 was not considered stochastically along with the calibrated source K_d value for Tc-99. Because these values are correlated, the calibration procedure used to develop the source K_d value would have to be modified to incorporate uncertainty in these parameters. Additionally, infiltration through the engineered cover while it remained intact was also treated as a fixed value.

The parametric uncertainty analysis presented here is not intended to be comprehensive because time and resources limited what could be accomplished in an uncertainty analysis for this project. Nevertheless, the analysis lays the framework for uncertainty analysis that can be refined later with revised parameter distributions and assumptions.

Parameter distributions used in the uncertainty analysis are presented in Table 22. Material properties and natural infiltration rates were largely taken from Rood (2000a) and used without modification. Distributions of partition coefficients were based on the data in Kincaid et al. (1998). Because partition coefficient values often times span an order of magnitude or more, log-triangular distributions were assumed. The mode of the distribution was taken to be the "best estimate" K_d value reported in Kincaid et al. (2000) which was also used in the deterministic simulations. The minimum K_d was taken to be the conservative estimate of the K_d as reported in Kincaid et al. (2000). This value was used in the Phase II screening described earlier in this report. The maximum of the distribution was taken to be the highest value reported in the range of possible K_d values in Kincaid et al. (1998). No distribution was assigned to the mobile fraction actinide K_d or the Tc-99 K_d (deterministic value of zero).

Uncertainty was also considered in the longevity of cover integrity. That is, the time in which the cover remains an effective infiltration barrier. The deterministic value for this

[3] Perl (Practical Extraction Reporting Language) is a scripting language available on most Unix workstations and recently made available for Microsoft Windows-based machines

parameter was 500 years. The distribution used in the stochastic simulation was assumed and was not based on engineering studies of the cover. The time over which the cover degraded was assumed to be equivalent to the time the cover remained intact. For example, if the cover remains intact for 300 years, then it degrades to natural infiltration in the next 300 years.

Table 22. Definition of Parameter Distributions used in the Uncertainty Analysis

Parameter	Distribution	Comments/Reference
Background infiltration (m y^{-1})	Triangular: minimum 0.0025; mode 0.005, maximum 0.01	Rood (2000a)
Longevity of cover integrity (yr)	Triangular: minimum 250, mode 500, maximum 750	Assumed
Longitudinal dispersivity in aquifer (m)	Triangular: minimum 13.75, mode 27.5, maximum 41.25	Rood (2000a)
Transverse dispersivity in aquifer (m)	Triangular: minimum 2.5, mode 5.0, maximum 7.5	Rood (2000a)
Darcy velocity in aquifer (m yr^{-1})	Truncated Lognormal: GM 32.9, GSD 2.33, minimum 3.0, maximum 250	Rood (2000a)
Bulk density, source unsaturated zone and aquifer (g cm^{-3})	Triangular: minimum 1.58, mode 1.97, maximum 2.36	Nominal values based on Kincaid et al (1998). Distribution based on Rood (2000a)
Aquifer porosity (m^3 m^{-3})	Triangular: minimum 0.097, mode 0.10, maximum 0.103	Rood (2000a)
Uranium K_d (mL g^{-1})	Log triangular: minimum 0.6, mode 3.0, maximum 79	Kincaid et al. (1998)
Thorium K_d (mL g^{-1})	Log triangular: minimum 40, mode 1000, maximum 2000	Kincaid et al. (1998)
Radium K_d (mL g^{-1})	Log triangular: minimum 8, mode 20, maximum 173	Kincaid et al. (1998)
Lead K_d (mL g^{-1})	Log triangular: minimum 2000, mode 6000, maximum 7900	Kincaid et al. (1998)
Carbon K_d (mL g^{-1})	Log triangular: minimum 0.25, mode 0.5, maximum 5.0	Kincaid et al. (1998)
Iodine K_d layers 5–13 (mL g^{-1})	Log triangular: minimum 0.3, mode 0.5, maximum 15	Kincaid et al. (1998)
Uranium solubility (mg L^{-1})	Triangular: minimum 1.0, mode 25, maximum 50	Rood (2000a)

The output variable presented in this report is the total drinking water dose at specific times after 1965. Output distributions of individual radionuclide concentrations and individual radionuclide doses are available electronically.

Distributions of model output were developed from 500 model realizations. The decision to use this number was based more on computer run time and disk storage considerations than statistical considerations. Although adding more realizations would result in greater confidence in the output distribution, the real question is what confidence do we have in any given percentile of the overall distribution. A confidence interval around percentiles of the output distribution was defined using a distribution-free approach developed in Hahn and Meeker (1991). The approach developed by Hahn and Meeker uses ordered statistics to define an interval where the true value of a given percentile lies at a specified level of confidence. In this way, confidence for any given percentile within the distribution could be defined. Of particular interest are the tails of the

distribution, because values at the tails (i.e., top and bottom) of the distribution change more with the number of model realizations; central values are more stable. The ordered statistics for the 5th and 95th percentiles for 500 model realizations are 25 and 475, respectively. That is, if the output values for 500 realizations are sorted in acceding order, the 5th percentile represents the 25th highest value; the 95th percentile represents the 475th highest value. The 95% confidence interval around the 5th percentile in terms of the ordered statistics is 15 and 35. The 95% confidence interval around the 95th percentile in terms of the ordered statistics is 465 and 485. We interpret this to mean we are 95% percent confident that 90% of the model output lies between the ordered statistics 15 and 485. The range of *values* represented by these ordered statistics will vary depending on the distribution. Distributions of model output were therefore expressed in terms of the 95% confidence interval around the 5th and 95th percentile values.

Uncertainty Analysis Results

A summary of the sampled parameters (Table 23) shows that median values of the assigned distributions were well represented by the sampled distributions. However, the tails of the distributions for some of the parameters were not well represented by the sampling. For example, the lower tail of the sampled distribution of uranium solubility was 2.85 mg L^{-1} but the assigned distribution had a minimum value of 1 mg L^{-1}. A greater number of model realizations and/or alternate distributions combined with Latin-Hypercube sampling may alleviate some of these sampling problems. Nevertheless, application of non-parametric confidence limits on the output distributions does account for some of the sampling error noted above and changes to the sampling scheme and assigned parameter distributions is left to a future iteration of this work.

Table 23. Statistics of the Sampled Parameter Distributions for 500 Model Realizations

Parameter	Minimum	Maximum	Mean	Median
Natural infiltration (m yr^{-1})	1.39E-03	1.91E-02	8.41E-03	7.88E-03
Cover longevity (years)	2.78E+02	7.37E+02	4.97E+02	4.99E+02
Longitudinal dispersivity (m)	1.58E+01	4.07E+01	2.77E+01	2.79E+01
Transverse dispersivity (m)	2.71E+00	7.44E+00	5.04E+00	5.02E+00
Darcy velocity in aquifer (m yr^{-1})	2.74E+00	2.25E+02	4.26E+01	3.21E+01
Bulk density (g cm^{-3})	1.61E+00	2.33E+00	1.96E+00	1.96E+00
Aquifer porosity (m3 m^{-3})	9.71E-02	1.03E-01	1.00E-01	1.00E-01
Uranium K_d, unsaturated layer 1 (mL g^{-1})[a]	8.63E-02	6.06E+01	5.91E+00	2.23E+00
Uranium K_d, unsaturated layer 2-13 (mL g^{-1})[a]	6.08E-02	4.27E+01	4.16E+00	1.58E+00
Thorium K_d, unsaturated layer 1 (mL g^{-1})[a]	3.99E+01	1.55E+03	4.62E+02	3.79E+02
Thorium K_d, unsaturated layer 2–13 (mL g^{-1})[a]	2.81E+01	1.09E+03	3.26E+02	2.67E+02
Radium K_d, unsaturated layer 1 (mL g^{-1})[a]	4.40E+00	1.34E+02	2.88E+01	2.11E+01
Radium K_d, unsaturated layer 2–13 (mL g^{-1})[a]	3.10E+00	9.43E+01	2.03E+01	1.49E+01
Lead K_d, aquifer (mL g^{-1})[a]	2.18E+03	9.68E+03	5.41E+03	5.27E+03
Iodine K_d, unsaturated layer 1 (mL g^{-1})[a]	1.71E-01	1.17E+01	1.61E+00	8.25E-01
Iodine K_d, unsaturated layer 2–13 (mL g^{-1})[a]	1.21E-01	8.24E+00	1.13E+00	5.81E-01
Carbon K_d, unsaturated layer 1 (mL g^{-1})[a]	2.30E-01	3.50E+00	8.73E-01	6.66E-01
Carbon K_d, unsaturated layer 2–13 (mL g^{-1})[a]	1.62E-01	2.47E+00	6.16E-01	4.69E-01
Uranium solubility (mg L^{-1})	2.85E+00	4.92E+01	2.54E+01	2.55E+01

[a] Sampled K_d value has been corrected for the percent gravel content in unsaturated layer 1 (17.3%) and unsaturated layers 2–13 (41.7%). Partition coefficients in the aquifer and source were assumed to be the same as in unsaturated layers 2–13.

Distributions of radionuclide concentrations at four output times (60 yrs, 800 yrs, 2000 yrs, and 10,000 yrs after 1965) are summarized in terms of four percentile values; 5th, 25th, 50th, and 95th percentile (Tables 24 and 25). Detailed output containing distributions of radionuclide concentrations for 28 separate output times are found in the ASCII files that accompany this report. The percentiles in Tables 24 and 25 do not include the 95% confidence intervals. The interval between the 5th and 95th percentiles spans upwards of 20 orders of magnitude or greater in some cases. Note that the span of the concentration distributions in year 60 for all mobile radionuclides (H-3, Tc-99, U-238MF, U-234MF, and Pu-239MF) were substantially smaller compared to other years of output. This difference is because the infiltration through the trenches and the cover was not considered stochastically and the radionuclides that dominate the dose at this time have K_d values fixed at zero.

Table 24. Percentiles of the Distribution of Groundwater Concentrations at 60 and 800 years from the Simulation Start Time (1965)

Radio-nuclide	Percentile, time = 60 years (Ci m^{-3})					Percentile, time = 800 years (Ci m^{-3})				
	5th	25th	50th	75th	95th	5th	25th	50th	75th	95th
H-3	2.00E-05	4.23E-05	7.02E-05	1.21E-04	2.71E-04	7.27E-24	1.95E-23	6.19E-23	3.52E-22	2.53E-21
C-14	7.07E-29	6.75E-25	1.19E-21	4.29E-19	1.20E-16	1.17E-27	2.25E-23	2.39E-20	4.47E-18	1.73E-15
I-129	2.77E-39	5.00E-32	5.93E-27	2.68E-23	2.89E-20	6.35E-37	1.58E-30	2.70E-25	1.45E-21	2.48E-18
Tc-99	1.24E-09	2.55E-09	4.37E-09	7.37E-09	1.71E-08	3.90E-10	1.09E-09	3.39E-09	1.86E-08	1.35E-07
U-238	1.29E-45	4.23E-37	7.64E-30	3.09E-24	3.95E-16	3.68E-43	7.13E-35	1.40E-28	2.75E-23	1.59E-15
U-234	1.84E-49	6.00E-41	1.16E-33	5.34E-28	1.02E-19	7.60E-46	1.52E-37	3.13E-31	6.27E-26	5.04E-18
Th-230	3.72E-57	2.24E-50	1.59E-44	3.18E-39	4.94E-32	9.63E-51	1.03E-43	4.33E-38	3.10E-33	4.39E-26
Ra-226	2.13E-54	1.35E-48	7.56E-43	5.21E-38	2.68E-31	1.62E-48	2.70E-42	1.80E-36	9.99E-32	7.06E-25
Pb-210	2.98E-57	2.43E-51	1.81E-45	9.55E-41	9.01E-34	3.59E-51	8.08E-45	4.86E-39	2.63E-34	1.88E-27
U-238MF	2.69E-10	5.66E-10	9.43E-10	1.64E-09	3.76E-09	7.65E-11	2.05E-10	6.41E-10	2.76E-09	1.23E-08
U-234MF	4.97E-11	1.05E-10	1.75E-10	3.03E-10	6.95E-10	1.42E-11	3.79E-11	1.18E-10	5.28E-10	2.26E-09
Pu-239MF	2.96E-11	6.28E-11	1.05E-10	1.83E-10	4.28E-10	4.45E-12	1.18E-11	3.56E-11	1.27E-10	4.57E-10

Table 25. Percentiles of the Distribution of Groundwater Concentrations at 2000 and 10,000 years from the Simulation Start Time (1965)

Radio-nuclide	Percentile, time = 2000 years (Ci m^{-3})					Percentile, time = 10,000 years (Ci m^{-3})				
	5th	25th	50th	75th	95th	5th	25th	50th	75th	95th
H-3	7.27E-24	1.95E-23	6.19E-23	3.52E-22	2.53E-21	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00
C-14	1.17E-27	2.25E-23	2.39E-20	4.47E-18	1.73E-15	2.65E-14	3.28E-10	4.92E-08	4.17E-07	1.58E-06
I-129	6.35E-37	1.58E-30	2.70E-25	1.45E-21	2.48E-18	2.30E-21	1.25E-14	2.10E-11	1.03E-09	5.00E-09
Tc-99	3.90E-10	1.09E-09	3.39E-09	1.86E-08	1.35E-07	5.56E-11	5.41E-10	2.50E-09	5.76E-09	1.73E-08
U-238	3.68E-43	7.13E-35	1.40E-28	2.75E-23	1.59E-15	5.81E-28	8.77E-19	8.52E-14	1.60E-09	8.19E-08
U-234	7.60E-46	1.52E-37	3.13E-31	6.27E-26	5.04E-18	1.97E-29	4.69E-20	7.55E-15	3.55E-10	2.67E-08
Th-230	9.63E-51	1.03E-43	4.33E-38	3.10E-33	4.39E-26	8.08E-34	1.33E-25	2.24E-20	7.77E-16	2.66E-13
Ra-226	1.62E-48	2.70E-42	1.80E-36	9.99E-32	7.06E-25	1.57E-30	9.89E-24	1.74E-18	3.05E-14	6.82E-12
Pb-210	3.59E-51	8.08E-45	4.86E-39	2.63E-34	1.88E-27	2.74E-33	5.08E-26	3.25E-21	8.14E-17	1.51E-14
U-238MF	7.65E-11	2.05E-10	6.41E-10	2.76E-09	1.23E-08	0.00E+00	2.02E-94	1.72E-66	1.03E-45	8.96E-16
U-234MF	1.42E-11	3.79E-11	1.18E-10	5.28E-10	2.26E-09	0.00E+00	3.64E-95	3.09E-67	1.85E-46	1.61E-16
Pu-239MF	4.45E-12	1.18E-11	3.56E-11	1.27E-10	4.57E-10	0.00E+00	2.48E-94	1.56E-67	3.56E-46	3.83E-16

Distribution of the total drinking water dose as a function of time (Figure 25) shows the uncertainty bounds increasing with increasing time. The uncertainty bounds (shaded area on Figure 25) represent the 5th and 95th percentiles with 95% confidence. Maximum doses during the 0–100 year time frame span about a factor of 25 between the 5th (<0.1 mrem yr^{-1}) and 95th (~ 2.5 mrem yr^{-1}) percentiles. Deterministic doses during the 0–100 year time frame are less than 5 mrem yr^{-1} and follow the median (50th percentile) of the distribution of predicted doses. Beyond 1000 years, the median estimate of the dose distribution does not exactly follow the deterministic value and the span of the 5th and 95th percentile values increase to over three orders of magnitude

at 10,000 years. Divergence of the median value from the deterministic dose estimate after 100 years is due to the increasing importance cover failure time and doses from sorbing radionuclides.

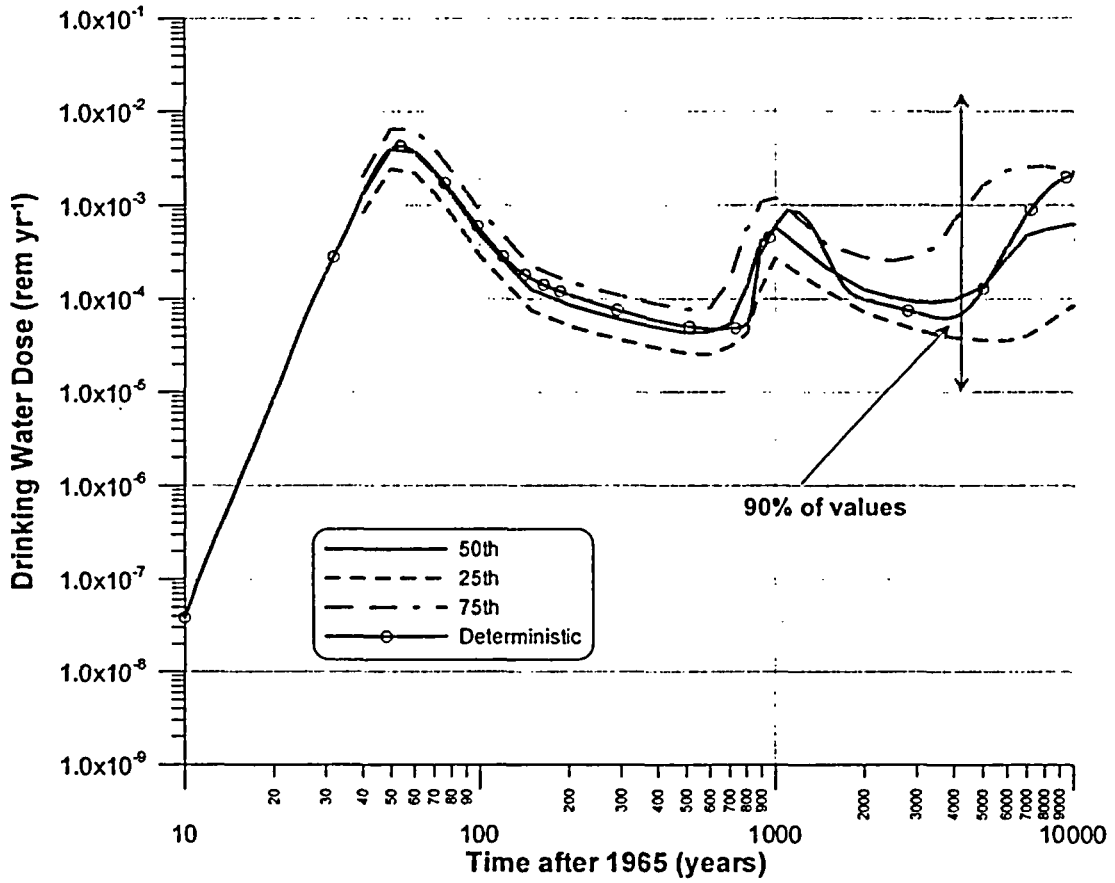

Figure 25. Stochastic simulation of the enhanced cover for closure in 2056 showing the distribution of total dose as a function of time. The shaded area represents the area between the 5th and 95th percentiles of the distribution (with 95% confidence). Also shown are the 25th, 50th and 75th percentiles of the distribution, and the deterministic results.

The results presented here indicate the precision of the model-predicted total drinking water dose is roughly a factor of 25 at times less than 100 years, and increases to over three orders-of-magnitude for times greater than 100 years. Results may be summarized by the following probability statements:

- We are 95% confident that there is a 95% probability that the *predicted* drinking water dose during the 0–100 year time frame will not exceed 17 mrem yr^{-1}.
- We are 95% confident that there is less than a 5% probability that the *predicted* drinking water doses during the 0–100 year time frame will exceed 17 mrem yr^{-1}.
- We are 95% confident that there is a 95% probability that the *predicted* drinking water dose during the 100–1,000 year time frame will not exceed 4.6 mrem yr^{-1}.
- We are 95% confident that there is less than a 5% probability that the *predicted* drinking water doses during the 100–1,000 year time frame will exceed 4.6 mrem yr^{-1}.

- We are 95% confident that there is a 95% probability that the *predicted* drinking water dose during the 1,000–10,000 year time frame will not exceed 28 mrem yr^{-1}.
- We are 95% confident that there is less than a 5% probability that the *predicted* drinking dose during the 1,000 –10,000 year time frame will not exceed 28 mrem yr^{-1}. Conversely, there is an equal probability that the dose during the 1,000–10,000 year time frame is less than 4.1×10^{-3} mrem yr^{-1}.

Note that the above statements only relate to the predicted doses and not to any real or actual doses.

Sensitivity Analysis

A quantitative sensitivity analysis was performed using the data generated during the uncertainty analysis. In the approach presented here, the Monte Carlo sampling techniques described earlier were used to propagate input parameter uncertainty into the predicted dose estimates. Then, using regression techniques, rank correlation coefficients were calculated between each parameter and the corresponding predicted dose. Parameter sensitivities are then established by the degree of correlation between the parameter and the output variable (predicted dose).

The methods used to evaluate parameter sensitivity are described in Crystal Ball software package (Decisioneering Inc. 2000). The rank correlation coefficients provide a quantitative measure of the sensitivity of the predicted dose to variations in the input parameters. Rank correlation replaces each input parameter and endpoint value pair, with its ranking within the distribution. Linear correlation of the rankings is then performed. Consider a simulation of n Monte Carlo trials where the parameters, a, b, and c are defined stochastically. The output variable defined as y, is calculated n times during the simulation. The results may be tabulated as follows:

$$
\begin{array}{llll}
a_1 & b_1 & c_1 & \Rightarrow \quad y_1 \\
a_2 & b_2 & c_2 & \Rightarrow \quad y_2 \\
a_3 & b_3 & c_3 & \Rightarrow \quad y_3 \\
\cdot & & & \\
\cdot & & & \\
\cdot & & & \\
a_n & b_n & c_n & \Rightarrow \quad y_n
\end{array}
$$

The subscript 1, 2, 3, ...n refer to the Monte Carlo trial number. To calculate the rank correlation coefficient, the values of a_i, b_i, c_i, and y_i are replaced by their ranking within the distribution of values. For example, suppose for the third Monte Carlo trial, the values a_3, b_3, c_3, are selected yielding an output value of y_3. Suppose 500 trials are performed and the value of a_3 was ranked at 23; –that is, it is the 23rd highest value within the distribution 500 values of a. The value of a_3 is replaced by 23. Likewise, the values of b_3, c_3, and y_3 are replaced by their respective ranks. Linear correlation is then performed between the ranks of each of the parameters and output variable, y.

The advantage of rank correlation over simple liner correlation is that it is nonparameteric. That is, it is not dependent on the underlying distribution of either the input or output variables. The rank correlation coefficient is given by (Press et al. 1992)

$$r_s = \frac{\sum_i \left(R_i - \overline{R}\right)\left(S_i - \overline{S}\right)}{\sqrt{\sum_i \left(R_i - \overline{R}\right)^2}\sqrt{\sum_i \left(S_i - \overline{S}\right)^2}} \tag{17}$$

where

r_s = the rank correlation coefficient
R_i = the rank of the input parameter value
S_i = the rank of the corresponding output value.

The advantage of using Monte Carlo techniques over that of a one-factor-at-a-time approach is that interactions between parameters are included in the analysis. For example, the sensitivity of the dose due to parameter Y may depend on the value chosen for parameter X. Rank correlation coefficients provides a meaningful measure of the degree to which parameters and the endpoint (drinking water dose) *change together*. The rank correlation coefficient takes on a value between –1 and +1. Perfect correlation is achieved when the absolute value of the correlation coefficient equals 1. Degree of correlation (and thereby degree of sensitivity), decreases with a decrease in the absolute value of the correlation coefficient. A positive correlation coefficient indicates that an increase in the value of the parameter results in an increase in the computational endpoint. A negative correlation coefficient indicates that an increase in the value of the parameter results in a decrease in the computational endpoint.

Another way to visualize the sensitivity analysis results is to compute the percent contribution each parameter has to the total variance. The contribution to the total variance was *approximated* using a simple technique described in the Crystal Ball® software (Decisioneering Inc. 2000) where the rank correlation coefficient for each parameter is squared and normalized to 100%. The output variable for this analysis is total (all nuclides) drinking water dose at specific times. Based on the results of the uncertainty analysis, four time-periods were chosen: 60, 800, 2000, and 10,000 years. These time periods correspond roughly to the times of maximum dose in the 0–10,000-year time frame.

Sensitivity Analysis Results

Results of the sensitivity analysis (Table 26) indicate that the sensitivity of a given parameter is time dependent. At 60 years after the start of operations, drinking water doses were most sensitive to Darcy velocity in the aquifer and to aquifer porosity. Correlation coefficients for most other parameters (excluding Darcy velocity and porosity) were not statistically significant, indicating there was no correlation between the drinking water dose and the parameter. Doses at 60 years were dominated by H-3.

At 800 years, drinking water doses were most sensitive to cover longevity, Darcy velocity in the aquifer, and background infiltration. The mean cover failure time was 500 years after installation of the cover in the year 2005 and therefore, failure would have a substantial impact on

non-sorbing radionuclides. Doses at 800 years were dominated by the mobile fraction of the uranium and plutonium isotopes and to a minor extent, Tc-99.

At 2000 years, drinking water doses were most sensitive to Darcy velocity in the aquifer and to background infiltration. Drinking water doses also show some sensitivity to the iodine and uranium K_d values as well as the uranium solubility. Doses at 2000 years were dominated by Tc-99, I-129, and U-238.

At 10,000 years, drinking water doses were most sensitive to uranium K_d values, background infiltration, Darcy velocity in the aquifer, and carbon and iodine K_d values. Doses at 10,000 years were dominated by U-238 and C-14.

Table 24. Rank Correlation Coefficient (RCC) and Percent Contribution to Variance for the Enhanced Cover with Closure in Year 2056

	60 years		800 years		2000 years		10,000 years	
Parameter	RCC	Percent variance	RCC	Percent variance	RCC	Percent variance	RCC	Percent variance
Background infiltration	1.54E-02	0.02%	3.27E-01	12.13%	1.58E-01	2.93%	4.09E-01	19.96%
Cover longevity	8.26E-03	0.01%	-7.03E-01	56.12%	2.46E-02	0.07%	-2.54E-01	7.69%
Longitudinal dispersivity	-8.62E-03	0.01%	1.31E-02	0.02%	-4.52E-02	0.24%	5.75E-02	0.39%
Transverse dispersivity	-2.41E-02	0.06%	6.42E-03	0.01%	-9.61E-03	0.01%	-1.73E-02	0.04%
Darcy velocity	-1.00E+00	94.59%	-5.01E-01	28.49%	-8.54E-01	85.64%	-3.48E-01	14.41%
Bulk density-source	2.98E-02	0.08%	-1.79E-02	0.04%	-3.72E-02	0.16%	8.55E-02	0.87%
Bulk density-unsat and aquifer	-9.97E-03	0.01%	1.71E-02	0.03%	6.84E-03	0.01%	3.58E-02	0.15%
Aquifer porosity	-1.07E-01	1.09%	-7.22E-02	0.59%	-7.48E-02	0.66%	-8.23E-02	0.81%
Uranium Kd, Unsat layers 1	-2.65E-02	0.07%	4.13E-03	0.00%	-9.75E-02	1.12%	-4.17E-01	20.70%
Uranium Kd, Unsat layer 2-13	-2.65E-02	0.07%	4.09E-03	0.00%	-9.74E-02	1.12%	-4.17E-01	20.70%
Thorium Kd, Unsat layers 1	6.75E-02	0.43%	-3.76E-02	0.16%	5.81E-02	0.40%	2.34E-02	0.07%
Thorium Kd, Unsat layers 2-13	6.75E-02	0.43%	-3.76E-02	0.16%	5.82E-02	0.40%	2.34E-02	0.07%
Radium Kd, Unsat layers 1	5.28E-02	0.26%	2.50E-02	0.07%	6.50E-02	0.50%	2.49E-02	0.07%
Radium Kd, Unsat layers 2-13	5.27E-02	0.26%	2.50E-02	0.07%	6.50E-02	0.50%	2.49E-02	0.07%
Lead Kd, aquifer	5.78E-02	0.32%	8.66E-02	0.85%	5.31E-02	0.33%	5.12E-02	0.31%
Iodine Kd, Unsat layers 1	-8.50E-02	0.68%	-6.61E-02	0.50%	-1.05E-01	1.30%	-1.72E-01	3.53%
Iodine Kd, Unsat layers 2-13	-8.50E-02	0.68%	-6.62E-02	0.50%	-1.05E-01	1.30%	-1.72E-01	3.53%
Carbon Kd, Unsat layers 1	-4.62E-03	0.00%	-2.02E-02	0.05%	-7.01E-02	0.58%	-1.66E-01	3.28%
Carbon Kd, Unsat layers 2-13	-4.57E-03	0.00%	-2.02E-02	0.05%	-7.01E-02	0.58%	-1.66E-01	3.27%
Uranium solubility	9.94E-02	0.93%	3.91E-02	0.17%	1.36E-01	2.18%	2.51E-02	0.08%

SUMMARY AND CONCLUSIONS

The analysis documented in this report was performed in response to the WDOH request to reconcile measured concentrations of radionuclides in the unsaturated zone with model-estimated values. Borehole measurements below trench 5 showed detectable concentrations of radionuclides. Evaluation of concentrations in the unsaturated zone required a new conceptual and mathematical model of waste disposal and radionuclide transport in the unsaturated zone. The conceptual and mathematical models incorporated into this assessment more accurately reflect the waste disposal history at the US Ecology site, time-variable infiltration as a consequence of pre-

and post-cover disposals, and radionuclide transport in the unsaturated zone. Three cover designs were evaluated; a site soils cover, an enhanced cover, and the US Ecology proposed cover. The benefit of installing an engineered cover as opposed to the site soils cover is a reduction of the predicted doses during the 100 to 600 year time period; the time the cover is assumed to remain intact. Doses after about 1,000 years are essentially the same for each cover design.

In addition to calculating aquifer concentrations and drinking water ingestion doses with estimates of uncertainty, the entire radionuclide inventory was re-examined for potential impacts to the groundwater pathway using a two-phase screening methodology. The screening results showed 15 radionuclides (C-14, Cl-36, H-3, I-129, Pu-238,-239,-240,-242, Ra-226, Tc-99, Th-230, Th-232, U-234, U-235, and U-238) were important to the groundwater pathway compared to the five radionuclides considered in the original DEIS (I-129, Tc-99, Cl-36, U-235, and U-238). However, the deterministic simulation results indicated only eight radionuclides contribute significantly to the drinking water ingestion dose for the 0 to 10,000-year time period. These radionuclides included four of the five radionuclides considered in the original DEIS plus H-3, C-14, U-234MF, and Pu-239MF. The immobile fractions of the actinides were estimated to arrive at the aquifer well after 10,000 years. Although Ni-63 and Sr-90 were screened from the initial radionuclide inventory, these radionuclides were detected in borehole samples and were therefore considered in the model calibration procedure. Conservative estimates of the drinking water dose estimates for Ni-63 and Sr-90 mobile fractions were less than 4 mrem yr^{-1}, and therefore these radionuclides did not warrant further consideration beyond model calibration. Tritium was shown to be the major dose contributor in the 0 to 100 year time period; however, deterministic doses were less than one 5 mrem yr^{-1}. Carbon-14 concentrations and doses were highest around 10,000 years but doses were less than a mrem yr^{-1}. With the possible exception of C-14 and H-3, the radionuclide screening performed in the original DEIS appears to have correctly chosen the major dose contributors.

Aquifer concentrations for these analyses were higher in some cases and lower in others compared to those reported in the original DEIS. The higher concentrations were mainly attributed to the assumed lifetime of the cover and inclusion of a mobile actinide fraction. In the original DEIS, the cover was assumed to last for infinity. This assumption is difficult to defend given that manmade disturbance could compromise the integrity of the cover anytime after institutional control. If cover integrity is assumed to be infinite, then estimated doses would be substantially lower. It is impossible to know how the cover will perform over long periods of time with no periodic maintenance or monitoring. For these reasons, a 500-year cover lifetime was assumed in the calculations.

Uranium-238 aquifer concentrations were higher in this assessment compared to the original DEIS despite the fact that the U-238 inventory in the original DEIS was more than an order of magnitude higher than in this analysis. This apparent discrepancy was attributed to the uranium solubility value, cover lifetime, and the mobile fraction of uranium. The solubility value used in the original deterministic DEIS calculations (1 mg L^{-1}) was on the lower end of the distribution of uranium solubility limits (~1 to 50 mg L^{-1}). If the original deterministic DEIS uranium solubility value (1 mg L^{-1}) were to be used with the current transport model, and the mobile fraction ignored, then U-238 concentrations would be about an order of magnitude lower than the values reported in this document.

Ancillary calculations using the HYDRUS code indicated that over time, the infiltration shadow beneath the cover would extend all the way to the aquifer. This is an important finding

and differs from the conservative assumption used in the original DEIS that the infiltration shadow only extended a minor distance below the bottom of the trench.

Uncertainty analysis indicated that the distribution of model-predicted doses spanned about a factor of 25 for times less than 100 years (5^{th} and 95^{th} percentiles). The span of the dose distribution increased to about three orders of magnitude at 10,000 years. The uncertainty analysis provides a measure of the precision of the model and should not be interpreted as the probability of any real or actual exposure occurring. It is simply a measure of the precision by which the model can estimate concentrations and doses far into the future.

The sensitivity analysis showed the cover lifetime to be a particularly sensitive parameter at 800 years from the start of the simulation in 1965. Doses were not particularly sensitive to its value at output times less than 200 years or greater than 2000 years.

Overall, the assessment integrates natural processes that govern the transport radionuclides in the subsurface, with known waste disposal histories, past operational practices, and future closure plans of the site into a transport model that estimates both past and future radionuclide migration from the US Ecology low-level radioactive waste site. Conservative assumptions were made where uncertainty exists and therefore, these results should be viewed as conservative estimates of radionuclide concentrations and drinking water doses.

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APPENDIX A

SUMMARY OF RADIONUCLIDE CONCENTRATIONS IN BORE HOLE SAMPLES

APPENDIX A: SUMMARY OF RADIONUCLIDE CONCENTRATIONS IN BORE HOLE SAMPLES

Table A-1 Summary of Measured Concentrations of Radionuclides in Boreholes Beneath Trench 5 (from US Ecology 1999, Appendix A)

Borehole	Depth (m)	U-238 (pCi/g)	U-235 (pCi/g)	U-234 (pCi/g)	Pu-239 (pCi/g)	Pu-238 (pCi/g)	Sr-90 (pCi/g)	Ni-63 (pCi/g)	Tc-99 (pCi/g)	Th-232 (pCi/g)	Ra-226 (pCi/g)
A	0.0	2.2E-01	1.3E-02	2.6E-01	<MDC	<MDC	3.0E-01	<MDC	<MDC	2.9E-01	2.9E-01
	3.0	3.2E-01	5.1E-02	4.6E-01	<MDC	<MDC	5.2E-01	5.0E+00	<MDC	5.3E-01	5.6E-01
	5.5	3.0E-01	3.4E-02	3.1E-01	<MDC	3.7E-02	2.3E-01	5.8E+00	<MDC	5.7E-01	5.0E-01
	8.2	3.0E-01	4.1E-02	3.3E-01	1.9E-02	<MDC	3.6E-01	2.9E+00	6.1E-01	1.2E+00	4.3E-01
	13.4	4.0E-01	4.5E-02	3.7E-01	1.9E-02	<MDC	3.5E-01	5.1E+00	<MDC	6.1E-01	5.4E-01
	16.2	3.1E-01	2.9E-02	3.7E-01	<MDC	<MDC	5.1E-01	6.1E+00	<MDC	7.1E-01	7.3E-01
	18.6	3.9E-01	2.4E-02	4.4E-01	3.6E-02	<MDC	1.4E-01	5.1E+00	<MDC	8.8E-01	5.1E-01
	21.3	4.1E-01	5.2E-02	3.9E-01	2.6E-02	<MDC	6.1E-01	3.5E+00	<MDC	5.4E-01	5.0E-01
B	0.0	3.8E-02	1.9E-02	3.2E-02	<MDC	<MDC	3.7E-01	2.0E+00	5.8E-01	2.3E-01	7.5E-01
	2.4	3.5E-02	mdc	2.6E-02	1.9E-02	<MDC	4.0E-01	5.0E+00	<MDC	6.1E-01	6.4E-01
	5.2	4.1E-02	1.9E-02	4.4E-02	1.9E-02	2.0E-02	6.0E-01	1.2E+00	<MDC	7.2E-01	5.2E-01
	7.9	4.0E-02	2.2E-02	6.4E-02	1.9E-02	<MDC	4.4E-01	4.5E+00	7.2E-01	5.5E-01	3.9E-01
	10.7	5.3E-02	2.9E-02	5.7E-02	1.9E-02	<MDC	8.4E-01	5.0E+00	<MDC	6.7E-01	4.9E-01
	13.1	6.8E-02	1.5E-02	9.7E-02	<MDC	2.0E-02	6.5E-01	4.5E+00	<MDC	7.2E-01	6.7E-01
	15.8	4.9E-02	7.0E-03	4.0E-02	<MDC	<MDC	1.1E+00	1.7E+00	<MDC	5.7E-01	2.9E-01
	18.3	4.8E-02	2.0E-02	1.0E-01	1.9E-02	<MDC	6.8E-01	1.5E+00	<MDC	5.4E-01	5.1E-01
	21.0	6.0E-02	1.4E-02	6.5E-02	<MDC	<MDC	3.4E-01	4.0E+00	<MDC	6.3E-01	6.4E-01
C	0.0	1.8E-01	1.1E-01	1.7E-01	<MDC	<MDC	2.1E-01	1.0E+01	<MDC	8.6E-02	5.0E-01
	3.0	1.5E-01	4.6E-02	1.7E-01	<MDC	<MDC	2.2E-01	3.7E+00	<MDC	1.4E-01	9.3E-01
	6.1	1.1E-01	3.7E-02	1.1E-01	<MDC	<MDC		3.9E+00	<MDC	2.8E-01	8.3E-01
	8.2	1.7E-01	2.4E-02	1.8E-01	<MDC	<MDC	1.2E-01	5.8E+00	6.9E-01	1.7E-01	5.9E-01
	11.0	1.8E-01	mdc	1.6E-01	<MDC	<MDC	8.9E-02	3.3E+00	<MDC	2.2E-01	8.5E-01
	13.4	2.9E-01	4.1E-02	2.7E-01	<MDC	<MDC		4.3E+00	<MDC	1.8E-01	7.2E-01
	16.2	2.0E-01	2.3E-02	1.9E-01	<MDC	<MDC	3.1E-01	5.3E+00	<MDC	1.7E-01	4.5E-01
	18.9	1.6E-01	mdc	2.1E-01	<MDC	<MDC		5.6E+00	<MDC	1.3E-01	6.6E-01
	21.3	2.0E-01	mdc	2.0E-01	<MDC	<MDC		4.3E+00	<MDC	2.3E-01	6.6E-01
	21.3	1.9E-01	2.4E-02	1.8E-01	<MDC	<MDC	7.2E-02	6.1E+00	<MDC	1.4E-01	3.3E-01
D	0.0	1.6E-01	5.2E-02	2.0E-01	<MDC	<MDC	1.6E-01	6.7E+00	<MDC	2.2E-01	5.3E-01
	2.4	1.2E-01	2.5E-02	1.3E-01	<MDC	<MDC	8.0E-02	4.2E+00	<MDC	2.2E-01	4.9E-01
	5.2	1.3E-01	mdc	1.5E-01	<MDC	<MDC	2.9E-01	4.1E+00	<MDC	1.8E-01	6.4E-01
	7.9	1.1E-01	mdc	1.5E-01	<MDC	<MDC	4.9E-02	6.4E+00	<MDC	2.4E-01	5.2E-01
	10.7	9.2E-02	mdc	1.4E-01	<MDC	<MDC	9.0E-02	3.4E+00	6.2E-01	1.7E-01	8.9E-01
	13.4	1.2E-01	mdc	1.3E-01	<MDC	<MDC	1.4E-01	6.7E+00	<MDC	1.6E-01	7.0E-01
	15.8	1.0E-01	mdc	1.5E-01	<MDC	<MDC	2.0E-01	6.5E+00	<MDC	7.2E-01	1.0E+00
	18.6	2.1E-01	mdc	3.3E-01	<MDC	<MDC	2.8E-01	4.0E+00	<MDC	2.4E-01	5.7E-01
	21.3	1.1E-01	mdc	2.3E-01	<MDC	<MDC	2.9E-01	5.7E+00	<MDC	2.4E-01	4.9E-01
	21.3	1.8E-01	2.8E-02	1.7E-01	<MDC	<MDC	3.2E-01	7.2E+00	<MDC	2.3E-01	6.4E-01

Table A-2 Mass of Uranium Isotopes in Bore Hole Samples and Computed Weight Percents

		Mass of isotope in 1 g of soil				Weight percent		
Borehole	Depth (m)	U-238	U-235	U-234	Total	U-238	U-235	U-234
A	0.0	6.53E-07	6.83E-09	4.12E-11	6.60E-07	98.9591%	1.0347%	0.0062%
	3.0	9.61E-07	2.68E-08	7.29E-11	9.88E-07	97.2788%	2.7138%	0.0074%
	5.5	8.86E-07	1.79E-08	4.88E-11	9.04E-07	98.0182%	1.9764%	0.0054%
	8.2	8.95E-07	2.15E-08	5.32E-11	9.17E-07	97.6437%	2.3505%	0.0058%
	13.4	1.18E-06	2.36E-08	5.86E-11	1.21E-06	98.0330%	1.9622%	0.0049%
	16.2	9.25E-07	1.52E-08	5.99E-11	9.40E-07	98.3727%	1.6209%	0.0064%
	18.6	1.18E-06	1.26E-08	6.97E-11	1.19E-06	98.9327%	1.0615%	0.0059%
	21.3	1.21E-06	2.73E-08	6.29E-11	1.24E-06	97.7942%	2.2008%	0.0051%
B	0.0	1.13E-07	9.98E-09	5.12E-12	1.23E-07	91.9022%	8.0937%	0.0042%
	2.4	1.04E-07		4.16E-12				
	5.2	1.22E-07	9.98E-09	7.05E-12	1.32E-07	92.4486%	7.5460%	0.0053%
	7.9	1.19E-07	1.16E-08	1.02E-11	1.31E-07	91.1610%	8.8312%	0.0078%
	10.7	1.58E-07	1.52E-08	9.13E-12	1.73E-07	91.2047%	8.7900%	0.0053%
	13.1	2.03E-07	7.88E-09	1.55E-11	2.11E-07	96.2529%	3.7398%	0.0074%
	15.8	1.46E-07	3.68E-09	6.41E-12	1.50E-07	97.5414%	2.4544%	0.0043%
	18.3	1.43E-07	1.05E-08	1.63E-11	1.54E-07	93.1529%	6.8365%	0.0106%
	21.0	1.79E-07	7.36E-09	1.04E-11	1.86E-07	96.0470%	3.9474%	0.0056%
C	0.0	5.34E-07	5.73E-08	2.79E-11	5.91E-07	90.3091%	9.6862%	0.0047%
	3.0	4.53E-07	2.42E-08	2.72E-11	4.78E-07	94.9339%	5.0604%	0.0057%
	6.1	3.34E-07	1.94E-08	1.76E-11	3.54E-07	94.4965%	5.4986%	0.0050%
	8.2	5.07E-07	1.26E-08	2.82E-11	5.20E-07	97.5684%	2.4262%	0.0054%
	11.0	5.43E-07		2.63E-11				
	13.4	8.65E-07	2.15E-08	4.29E-11	8.87E-07	97.5656%	2.4296%	0.0048%
	16.2	6E-07	1.21E-08	2.98E-11	6.12E-07	98.0196%	1.9756%	0.0049%
	18.9	4.65E-07		3.4E-11				
	21.3	6.09E-07		3.12E-11				
	21.3	5.76E-07	1.26E-08	2.82E-11	5.88E-07	97.8520%	2.1432%	0.0048%
D	0.0	4.86E-07	2.73E-08	3.16E-11	5.14E-07	94.6741%	5.3198%	0.0061%
	2.4	3.64E-07	1.31E-08	2E-11	3.77E-07	96.5113%	3.4834%	0.0053%
	5.2	3.88E-07		2.4E-11				
	7.9	3.4E-07		2.37E-11				
	10.7	2.74E-07		2.16E-11				
	13.4	3.55E-07		2.15E-11				
	15.8	3.04E-07		2.35E-11				
	18.6	6.38E-07		5.25E-11				
	21.3	3.34E-07		3.7E-11				
	21.3	5.25E-07	1.47E-08	2.71E-11	5.40E-07	97.2693%	2.7257%	0.0050%

FINAL REPORT

FOLAT: A Model for Assessment of Leaching and Transport of Radionuclides in Unsaturated Porous Media

January, 2003

Submitted to Washington State Department of Health
in partial fulfillment of contract No. N10996

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FINAL REPORT

FOLAT: A Model for Assessment of Leaching and Transport of Radionuclides in Unsaturated Porous Media

January, 2003

Arthur S. Rood

K-Spar Inc. *Scientific Consulting*

ABSTRACT

The FOLAT (First-Order-Leach-And-Transport) model was developed to estimate radionuclide fluxes to the aquifer from radionuclide sources either in a waste disposal facility or present in the form of residual contamination in soils from various industrial processes. The model was designed to be simple yet robust, incorporating many of the major processes and assumptions that typically determine the results of prospective radiological assessments. It was not intended to be a predictive tool. The subsurface environment is assumed to be composed of a series of "compartments". Within each compartment, radionuclides enter, mix, sorb, decay, and are eventually removed by the downward movement of water. Each compartment may have its own unique qualities that include horizontal and vertical dimensions, bulk density, porosity, hydraulic conductivity, net water flux through the compartment, and sorptive properties. The model is formulated as a series of ordinary differential equations that are solved through the use of a 4th order Runga Kutta routine. Time variable radionuclide release rates and water fluxes through each compartment is easily incorporated into a simulation. Solubility controls are imposed on radionuclide concentrations in pore water in each of the compartments. The model can handle up to a six-member decay chain. Radioactive progeny move according to their own transport properties.

The model is coded in FORTRAN and input is provided through three ASCII files. Code output includes pore water concentrations, compartment inventories, and radionuclide fluxes through each of the compartments. The code was verified using analytical solutions to the governing equations and benchmarked against two other radionuclide transport codes that incorporate the advection-dispersion transport equation. Benchmark exercises indicate that FOLAT provides aquifer fluxes comparable to those obtained from the advection-dispersion equation.

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INTRODUCTION

Evaluation of radionuclide transport in the unsaturated zone is both a complex and evolving science. Typically, one is presented with the problem of assessing the release and transport of radionuclides to the aquifer from radionuclide sources either in a waste disposal facility or present in the form of residual contamination in soils from various industrial processes. Many models ranging from the simple to complex have been developed to address this problem. Several models specifically designed for this task include the Disposal Unit Source Term Model (DUST) (Sullivan 1996), RESRAD (Yu et al. 2000), and SESOIL (Scott and Hetrick 1994) for non-radiological contaminants. Others may rely on models designed to solve the general governing equations of fluid flow and transport in a variable saturated porous media. Several examples of such models include STOMP (PNNL 1996) and PORFLOW (ACRI 1996). Oftentimes, the available characterization data for an assessment are poor which limits the effectiveness of a complex model. Furthermore, many of assessments are prospective and entail model predictions out to tens of thousands of years. The reliability of any model is questionable under such conditions and by their very nature, such model predictions cannot be confirmed with field observations. The nature of the prospective analysis and the complexity of problem oftentimes leads to simplifying but conservative assumptions about radionuclide release and transport in the subsurface environment. The term conservative is used in the context of providing overestimates of radionuclide concentrations at points where exposure to humans may be possible and thereby ensuring radiological impacts to any real person will not exceed the calculated values.

This report describes the FOLAT (First-Order-Leach-And-Transport) model, which was designed to assess radionuclide leaching and transport in an unsaturated porous media for the purposes of low-level radioactive waste performance assessment. The model may also have other applications. Its primary output is radionuclide fluxes from the base of the unsaturated zone. This is typically the surface of an aquifer. The model employs the soil leaching model proposed by Baes and Sharp (1983) and extends that model to multiple subsurface layers with potentially differing transport properties. The model was designed to incorporate readily available data and the dominant processes that tend to impact the results of such assessments. Such processes include time-variable water infiltration rates and spatially variable equilibrium sorption. Estimated water fluxes in the unsaturated zone from models such as UNSAT-H (Fayer 2000), may be incorporated into the FOLAT model.

CONCEPTUAL MODEL

The conceptual model for FOLAT is relatively simple (Figure 1). The subsurface environment is envisioned to be composed of a series of "compartments". Within each compartment, radionuclides enter, mix, sorb, decay, and are eventually removed by the downward movement of water. Each compartment may have its own unique qualities that include horizontal and vertical dimensions, bulk density, porosity, hydraulic conductivity, net water flux through the compartment, and sorptive properties. Water flux through each compartment may change as a function of time. As the water flux changes, so too does the moisture content of the compartment. Radionuclides sorb on to the solid matrix as described by the equilibrium partitioning coefficient or *Kd*. Sorption retards the overall downward movement of radionuclides. The rate of transport of radioactive decay products or progeny that form during vertical transport

of a parent radionuclide are governed by the sorptive properties of the progeny, and not those of the parent.

Radionuclides may be present in each of the compartments at the start of the simulation, or alternatively, the parent member of the decay chain may be placed over time in the uppermost compartment. Concentrations of radionuclides in pore water are not allowed to exceed their solubility limit. Unit gradient conditions are assumed to apply to each compartment.

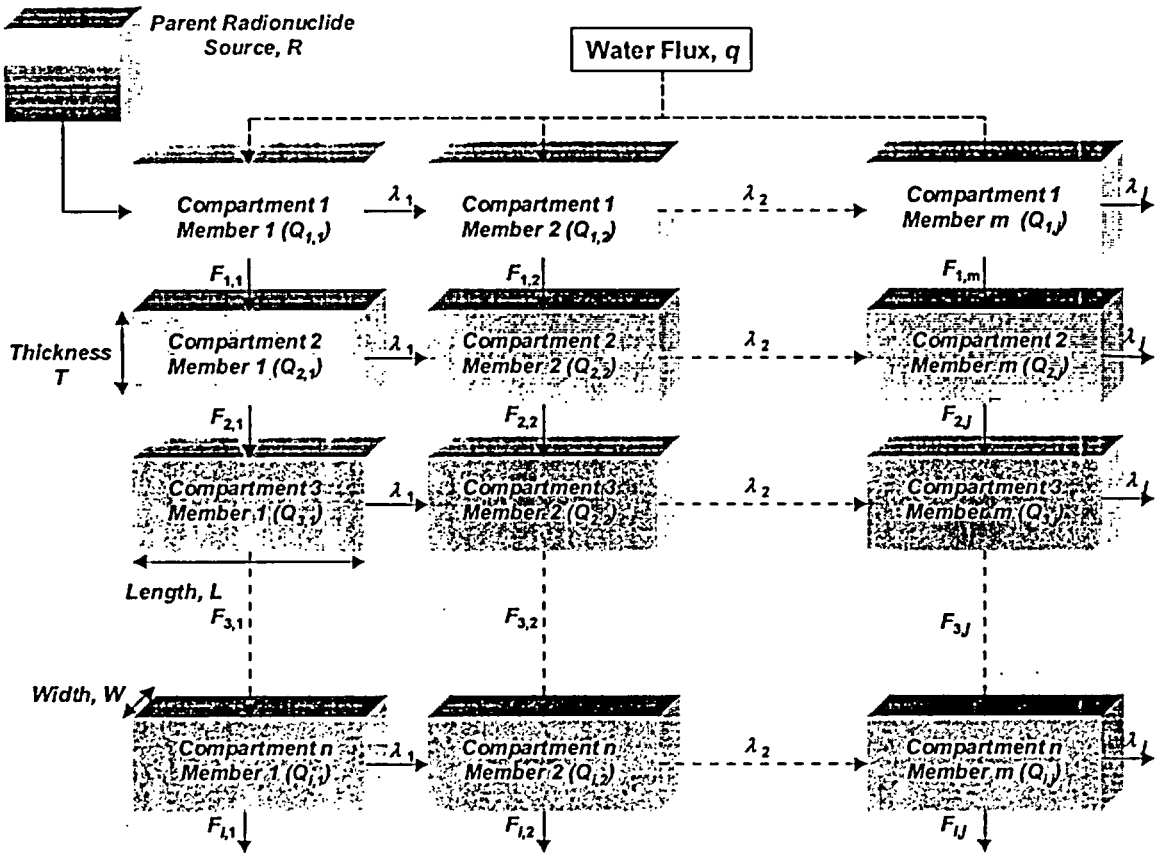

Figure 1. Conceptual model for FOLAT. In this illustration, there are n compartments for a decay chain consisting of m members. The variable $F_{i,j}$, is the removal rate (flux) from compartment i for decay chain member j. The variable, λ_j is the decay rate constant for decay chain member j. The variable $Q_{i,j}$ represents the number of atoms of decay chain member j in the i^{th} compartment. The water flux (q indicated by the dashed line) is shown entering the first compartment but is also an input for all remaining compartments. Arrows connecting each compartment indicate the direction of radionuclide transport. Dotted arrows indicate other compartments or decay chain members may be present.

MATHEMATICAL MODEL

Ordinary differential equations describe the mass balance of radionuclides in each of the compartments. Radionuclide concentrations in pore water and the radionuclide flux from each

compartment are determined from the radionuclide inventory within each compartment. The uppermost or first compartment for the first (parent) member of the decay chain is described by

$$\frac{dQ_{1,1}}{dt} = R(t) - F_{1,1}(t) - \lambda_1 Q_{1,1} \quad (1)$$

where

$Q_{1,1}$ = the number of atoms in compartment 1 for decay chain member 1 (atoms)
$R(t)$ = the input rate of decay chain member 1 into compartment 1 (atoms time^{-1})
$F_{1,1}(t)$ = the removal rate (flux) of decay chain member 1 from compartment 1 to compartment 2 (atoms time^{-1})
λ_1 = the decay rate constant for decay chain member 1 (time^{-1}).

For simplicity and clarity, all equations are written in terms of the number of atoms of each decay chain member. The mass balance equation for the remaining compartments is given by

$$\frac{dQ_{i,j}}{dt} = F_{i-1,j}(t) - \lambda_j Q_{i,j} + \lambda_{j-1} Q_{i,j-1} - F_{i,j}(t) \quad (2)$$

where i is the index for the compartment and j is the index for the decay chain member and $i \neq 1$ and $j \neq 1$. Other terms are defined previously. Equation 2 assumes the branching ratio between the parent and radioactive progeny is 1.0 (i.e. 100% of the parent decays to the progeny). When the radionuclide concentration in pore water is less than the solubility limit, the flux term in Equations 1 and 2 is given by

$$F_{i,j}(t) = \left(\kappa_{i,j}(t) + \eta_{i,j}\right) Q_{i,j} \quad (3)$$

where

$\kappa_{i,j}(t)$ = the leach rate constant for compartment i and decay chain member j (time^{-1})
$F_{i,j}(t)$ = the flux of decay chain member j from compartment i into compartment i+1 (atoms time^{-1})
$\eta_{i,j}$ = a fixed removal rate constant for compartment i and decay chain member j (time^{-1}).

In general, only the leach rate constant is used to remove radionuclides from a compartment. However, the user may which to bypass this calculation and put in their own removal rate constant. We have included $\eta_{i,j}$ for this situation. When the pore water concentration exceeds the solubility limit, then the flux term in Equations 1 and 2 is given by

$$F_{i,j}(t) = S_j \, q_i(t) \, L_i \, W_i \quad (4)$$

where

S_j = the solubility limit of decay chain member j (atoms m^{-3})
$q_i(t)$ = water flux through compartment i as a function of time (m time^{-1})
L_i = length of compartment i (m)

W_i = width of compartment i (m).

In Equations 3 and 4, $i \leq n$, where n is the number of compartments in the simulation. Likewise, $j \leq m$, where m is the number of decay chain members including the parent. The pore water concentration in compartment i for decay chain member j ($C_{i,j}$) is given by

$$C_{i,j}(t) = \frac{Q_{i,j}(t)}{\theta_i(t) L_i W_i T_i \left(1 + \frac{Kd_{i,j}\rho_i}{\theta_i(t)}\right)} \quad (5)$$

where

$\theta_i(t)$ = volumetric moisture content in compartment i as a function of time ($m^3\ m^{-3}$)
$Kd_{i,j}$ = equilibrium partition coefficient for compartment i and decay chain member j ($mL\ g^{-1}$)
ρ_i = bulk density of compartment i ($g\ mL^{-1}$)
T_i = thickness of compartment i (m)
L_i = length of compartment i (m)
W_i = width of compartment i (m).

The term, $1 + Kd_{i,j}\ \rho_i/\theta(t)_i$ is the retardation coefficient and is 1.0 for a Kd of zero. The leach rate constant is given by Baes and Sharp (1983) as

$$\kappa_{i,j}(t) = \frac{q_i(t)}{\theta_i(t)\ T_i \left(1 + \frac{Kd_{i,j}\rho_i}{\theta_i(t)}\right)} \quad (6)$$

and the decay rate constant is given by

$$\lambda_j = \frac{\ln(2)}{T1/2_j} \quad (7)$$

where $T1/2_j$ = half-life of decay chain member j.

Note that $\kappa_{i,j}$ can be set to zero by letting $q_i(t) = 0$ or setting $Kd_{i,j}$ to a sufficiently large value such that $\eta_{i,j} >> \kappa_{i,j}$.

The value q/θ represents the average linear water velocity (pore velocity) through the compartment and is based on the unit gradient model. The unit gradient model assumes water infiltration is driven primarily by gravity flow. Darcy's law in a one dimensional, vertically aligned, unsaturated soil column is given by

$$q = K(\theta)\left(\frac{\partial H}{\partial z} + \frac{\partial \psi}{\partial z}\right) \quad (8)$$

where

H = elevation head (m),
ψ = suction or pressure head (m),

$K(\theta)$ = unsaturated hydraulic conductivity of the column (m time^{-1})
z = length of the column (m).

Under unit gradient conditions, $\partial\psi/\partial z = 0$, $\partial H/\partial z = 1$, and $\partial\theta/\partial t = 0$. Therefore, $q = K(\theta)$. The unsaturated hydraulic conductivity is a function of the volumetric moisture content of the media. The volumetric moisture content is the fraction of the bulk media that is filled with water. When a porous media is saturated (i.e., all the pore spaces are filled with water), the volumetric moisture content is equal to the effective porosity of the media. It is assumed that unit gradient conditions apply to all compartments in the model. Moisture content changes in response to changes in the net water flux through a compartment are assumed to equilibrate rapidly compared to the rate of transfer of radionuclides among compartments. The relationship between hydraulic conductivity and moisture content differs for different rock types. The volumetric water content for a given material and a constant water flux is usually determined using equations that have been fitted to field data relating suction head to volumetric moisture content and hydraulic conductivity. The unsaturated hydraulic conductivity can be written in terms of the moisture content and soil fitting parameters, α and n, described in van Genuchten (1978) and given in Equations 9 and 10.

$$K(\theta) = K_{sat}\left(\frac{\theta - \theta_r}{\theta_s - \theta_r}\right)^{1/2}\left\{1-\left[1-\left(\frac{\theta-\theta_r}{\theta_s-\theta_r}\right)^{\frac{1}{m}}\right]^{m}\right\}^{2} \tag{9}$$

and

$$\left(\frac{\theta-\theta_r}{\theta_s-\theta_r}\right)=\left(\frac{1}{1+\alpha n}\right)^{m} \tag{10}$$

where
θ = volumetric moisture content (m^3 m^{-3}),
θ_r = residual moisture content (m^3 m^{-3}),
θ_s = saturated moisture content (m^3 m^{-3}),
K_{sat} = saturated hydraulic conductivity (m y^{-1}),
α = fitting parameter (m^{-1}),
n = fitting parameter,
m = 1 - 1/n.

CODE IMPLEMENTATION

Equations 1-10 are coded into a FORTRAN program that 1) reads user input from ASCII files, 2) computes unit conversions and pore water concentrations, 3) solves Equations 1 and 2, and 4) writes output to ASCII files. Equations 1 and 2 were solved using a 4th-order Runga Kutta ordinary differential equation solver from Press et al. (1992). Input to the code is through three ASCII files. The primary input file is termed the parameter definition file and defines model options, compartment properties, radionuclide parameter values, and initial inventories of radionuclides in each compartment. A second file (listed in the parameter definition file) contains the net water fluxes as a function time for each compartment. A third file contains the parent

radionuclide release rates to the first (uppermost) compartment. Output includes a general output file that echoes back the user input, leach rate constants and pore water concentrations at the start of the simulation, and pore water concentrations, radionuclide flux, and compartment inventories as a function of time for each output time defined in the parameter definition file. File structure for each of the three input files is described in Tables 1–3. All input files are free-form ASCII, which may be created in any standard text editor. Each card in the parameter definition file represents one or more lines of input. The code ignores lines where a dollar sign ($) in placed in the first column, thereby facilitating comments in the input file. Suggested default values (when applicable) are put in the description column of Table 1 in parentheses.

Radionuclide inventories and release rates are input in units of Curies. These quantities are converted to atoms using

$$Q_{mass} = \frac{Q_{activity}\left(3.7\times10^{10}\,\mathrm{dps\,Ci^{-1}}\right)\left(3.1536\times10^{7}\,\mathrm{s\,y^{-1}}\right)MW}{\lambda N_A} \tag{11}$$

where

MW = molecular weight (g mol^{-1})
N_A = Avogadro's number (6.023×10^{23})
dps = number of nuclear disintegrations per second
Q_{mass} = number of atoms of a radionuclide in a given compartment (atoms)
Q_{mass} = activity of a radionuclide in a given compartment (Ci)

Release rates are read into the code via an ASCII file containing pairs of time-release rate values. Values at times in between the tabulated values are linearly interpolated. A similar procedure applies to water fluxes.

Water fluxes through each compartment as a function of time are read into the code via an ASCII file. Values at times in between the tabulated values are linearly interpolated. Under steady –state conditions, the water fluxes through each compartment are the same. Under transient conditions, water flux may change as a function of time and space. Unsaturated flow models (such as UNSAT-H) may be used to compute these water fluxes for input into FOLAT. Alternatively, a FOLAT-compatible water flux file may be generated using the preprocessor described in Appendix A.

The code was compiled using Version 5.60a of the Lahey, LF95 FORTRAN compiler for both Microsoft Windows® 9x and Linux operating systems. Run times on an Athlon 760 mHz processor running Linux Mandrake 7.0 were less than a second for the verification and test problems presented in this document.

Code versions are identified by the version date which is printed in all the output files. Furthermore, headers in each of the subroutines describe the purpose of the subroutine, variables passed, the units (if applicable), and a version history.

Table 1. Parameter Definition File for the FOLAT Program

Card	Code variable	Type/format	Units	Description
1	Title	CHAR/A80		Title of run
2	Fileout	CHAR/A60		Output file name
3	Fileppt	CHAR/A60		File containing net water flux in each compartment as a function of time
4	Filerel	CHAR/A60		File containing radionuclide fluxes to the first compartment for the parent nuclide
5	eps	REAL/*		Required accuracy of solution (1×10^{-6})
5	h1	REAL/*	year	Beginning time step (0.0001 year)
5	hmin	REAL/*	year	Minimum time step (1×10^{-30} year)
6	mlayer	INT/*		Number of compartments in the simulation (maximum = 50)
6	nprog	INT/*		Number of decay chain members (maximum = 6)
6	nmat	INT/*		Number of material types (must be ≤ mlayer)
7	cname(*i*)	CHAR*6/*		Name of each member of the decay chain, nprog values are read from a single line. This name is concatenated with ".rel" extension to for the file name where fluxes from the last compartment are output.
8	mw(*j*)	REAL/*	g mol^{-1}	Molecular weight of each decay chain member, nprog values are read from a single line.
9	sol(*j*)	REAL/*	mg L^{-1}	Solubility limit of each decay chain member, nprog values are read from a single line.
10	thalf(*j*)	REAL/*	years	Half life of each decay chain member, nprog values are read from a single line.
NOTE: Card 11 is read nprog times				
11	y(*i*,*j*)	REAL/*	Ci	Initial activity in each compartment for decay chain member *j*. mlayer number of values are read from each line.
NOTE: Card 12 is read nprog times				
12	kd(*i*,*j*)	REAL/*	mL g^{-1}	Equilibrium sorption coefficient for each compartment for decay chain member *j*. mlayer number of values are read from each line.
NOTE: Card 13 is read nprog times				
13	kx(*i*,*j*)	REAL/*	y^{-1}	Alternative removal rate constant for decay chain member *j*. mlayer number of values are read from each line.
NOTE: Card 13a, 13b, and 13c define the compartment properties and are read nmat number of times. Compartments must be defined in ascending order				
13a	h	INT/*		Beginning compartment number to define compartment properties.
13a	j	INT/*		Ending compartment number to define compartment properties.
13b	thick(*i*)	REAL/*	m	Thickness of compartment i
13b	rho(*i*)	REAL/*	g mL^{-1}	Bulk density of compartment i
13b	lth(*i*)	REAL/*	m	Length of compartment i
13b	width(*i*)	REAL/*	m	Width of compartment i
13c	sk(*i*)	REAL/*	m y^{-1}	Saturated hydraulic conductivity of compartment i
13c	ths(i)	REAL/*	m^{-3} m^{-3}	Saturated porosity of compartment i

Table 1. Parameter Definition File for the FOLAT Program

Card	Code variable	Type/format	Units	Description
13c	thr(*i*)	REAL/*	$m^{-3} m^{-3}$	Residual moisture content of compartment i
13c	alpha(*i*)	REAL/*	m^{-1}	van Genuchten fitting parameter for compartment i
13c	rn(*i*)	REAL/*		van Genuchten fitting parameter for compartment i
14a	ntimes	INT/*		number of output time periods
NOTE: Card 14b is read ntime number of times				
14b	t1	REAL/*	years	Begin time of output
14b	t2	REAL/*	years	End time of output
14b	tp	REAL/*	years	Print step

Table 2. Description of the Water Flux Input File

Line number	Code variable	Description
1	Junk	Column header (discarded)
2 to $n+1$[a]	precip(*k*,1)	Time in years from the start of the simulation for the k^{th} record
2 to $n+1$[a]	precip(*k*,i)	Water flux ($m\ y^{-1}$) for total number of compartments in the simulation for the k^{th} time record

[a] *n* is the number of time and water flux records. A minimum of two records is needed to operate the code.

Table 3. Description of the Radionuclide Release Rate Input File

Line number	Code variable	Description
1	Junk	Column header (discarded)
2 to $n+1$[a]	rel(*k*,1)	Time in years from the start of the simulation for the k^{th} record
2 to $n+1$[a]	rel(*k*,2)	Radionuclide release rate ($Ci\ y^{-1}$)for the k^{th} record

[a] *n* is the number of time, radionuclide release rate records. A minimum of two records is needed to operate the code.

Code Execution

Execution of FOLAT is performed on the command line by typing

[*path*] FOLAT [*parameter definition file*]

where *path* is the full pathname to the executable file. If the parameter definition file is omitted, then the program will look for the default parameter file named, FOLAT.PAR. If this file is not found, the program will abort.

CODE VERIFICATION

Code verification is defined here as confirmation that the model was written and implemented in the computer code correctly. To do this, quantities output from the FOLAT code (such as pore water concentrations, fluxes, and radionuclide inventories) were compared with like quantities calculated using other codes or analytical expressions. Four test problems were defined,

which compared these quantities with output from analytical solutions to the governing equations where solutions exist. In one case (Verification Problem 4), output from another code implementing a similar solution is used. Analytical expressions are developed first, followed by presentation of the test problems and results. Differences between FOLAT calcuations and analytical solutions to the test problems are expressed as the percent difference (%*d*) as given by

$$\%d = \frac{V_a - V_f}{V_a} \times 100 \tag{12}$$

where V_a = the value calculated by analytical solution, and V_f = the value calculated by FOLAT. The FOLAT output files are presented in Appendix B.

Analytical Solutions for Governing Equations

Equation 1 has three solutions depending on the initial conditions and whether there is an external radionuclide source involved. For the case where $Q_{1,1} = Qo_{1,1}$ at $t = 0$, $\eta_{1,1} = 0$, $R(t) = 0$ for all time, and $C_{11} \leq S_1$, the solution is

$$Q_{1,1}(t) = Qo_{1,1} \exp\left(-\left(\kappa_{1,1} + \lambda_1\right)t\right) \tag{13}$$

For the case where $Q_{1,1} = 0$ at $t = 0$, $\eta_{1,1} = 0$, and $R(t)$ is a constant, R, for all time, the solution is

$$Q_{1,1}(t) = \frac{R}{\kappa_{1,1} + \lambda}\left(1 - \exp\left[-\left(\kappa_{1,1} + \lambda_1\right)t\right]\right) \tag{14}$$

The solution to the second compartment (Equation 2) for the conditions in Equation 14 and $Q_{2,1} = 0$ at t = 0 can be found using Laplace transforms. The governing differential equation is given by

$$\frac{dQ_{2,1}}{dt} = \frac{\kappa_{1,1} R}{\gamma}\left[1 - \exp(-\gamma t)\right] - \delta Q_{2,1} \tag{15}$$

where

$\gamma = \kappa_{1,1} + \lambda_1$

$\delta = \kappa_{2,1} + \lambda_1$.

The solution is

$$Q_{2,1}(t) = \kappa_{1,1} R\left(\frac{1}{\gamma\delta} + \frac{\gamma e^{-\delta t} - \delta e^{-\gamma t}}{\gamma\delta(\delta - \gamma)}\right) \tag{16}$$

For a decay chain of more than one member, a generalized solution to the multiple compartment problem has been developed by Birchall (1986) for the limiting conditions, $R(t) = 0$ for all times, $d\kappa/dt = 0$, $dq/dt = 0$, and $Q_{i,j} = Qo_{i,j}$ at $t = 0$.

When the pore water concentration exceeds the solubility limit of the radionuclide, then a solubility-limited release is calculated. The solubility-limited release in the first compartment where $R(t) = 0$ for all times and $\eta_{1,1} = 0$ is described by the differential equation

$$\frac{dQ_{1,1}}{dt} = -\lambda_1 Q_{1,1} - R_s \tag{17}$$

where

$R_s = S_1 \times q(t)_1 \times L_1 \times W_1$.

The solution to Equation 17 for the initial conditions, $Q_{1,1}=Qo_{1,1}$ at $t = 0$, is

$$Q_{1,1}(t) = Qo_{1,1} \exp(-\lambda_1 t) - \frac{R_s}{\lambda_1}[1 - \exp(-\lambda_1 t)] \tag{18}$$

When the pore water concentration is less than the solubility limit, then $Q(t)$ is described by

$$Q_{1,1}(t) = Q_{1,1}(t_{s1}) \exp[-(\kappa_{1,1} + \lambda_1)(t - t_{s1})] \tag{19}$$

where $t \geq t_{s1}$, and t_{s1} is the time when the pore water concentration drops below the solubility limit.

Verification Problem 1

Verification Problem 1 considers a three-compartment model with an initial radioactivity inventory of zero in all compartments and a constant release rate of 1 Ci y^{-1} into the first compartment. A hypothetical radionuclide having a half-life of 100 years and a molecular weight of 138 g mol^{-1} is simulated. Water flux is assumed to be at steady state and different for each compartment. Input data are presented in Table 4. Radionuclide inventories were converted to pore water concentrations using Equation 5. Radioactive inventories were calculated using Equation 14 for the first compartment and Equation 16 for the second compartment.

Table 4. Parameter Values used in Verification Test Problems 1, 2, and 4

Parameter	Compartment 1	Compartment 2	Compartment 3
Thickness (m)	1.00E+00	1.00E+00	1.00E+00
Length (m)	1.00E+01	1.00E+01	1.00E+01
Width (m)	1.00E+01	1.00E+01	1.00E+01
Bulk Density (g cm^{-3})	1.50E+00	1.50E+00	1.50E+00
Saturated hydraulic conductivity (m y^{-1})	1.71E+03	1.71E+03	1.71E+03
Total porosity (m^3 m^{-3})	2.72E-01	2.72E-01	2.72E-01
Residual moisture content (m^3 m^{-3})	3.21E-02	3.21E-02	3.21E-02
van Genuchten alpha (m^{-1})	7.51E+00	7.51E+00	7.51E+00
van Genuchten *n*	2.30E+00	2.30E+00	2.30E+00
Water flux (m y^{-1})	1.00E-01	5.00E-02	2.50E-02

Half-life (y)	1.00E+02	1.00E+02	1.00E+02
Molecular weight (g mol^{-1})	1.38E+02	1.38E+02	1.38E+02
Solubility limit (mg L^{-1})	1.00E+00	1.00E+00	1.00E+00
Partition coefficient (mL g^{-1})	1.00E-01	5.00E-01	1.00E+00
Initial moisture content[a]	6.06E-02	5.61E-02	5.24E-02
Leach rate constants[a] (y^{-1})	4.75E-01	6.20E-02	1.61E-02
Solubility limit[a] (Ci m^{-3})	2.59E+01	2.59E+01	2.59E+01

[a] Calculated values

Results of Verification Problem 1 (Table 5) for the first two compartments indicate good agreement between FOLAT and the analytical solution. The differences are ≤0.004%.

Table 5. Results of Verification Problem 1

	Compartment 1						Compartment 2		
Time (y)	FOLAT Inventory (Ci)	Analytical solution (Ci)	% Difference	FOLAT Conc. (Ci m^{-3})	Analytical solution (Ci m^{-3})	% Difference	FOLAT Inventory (Ci)	Analytical solution (Ci)	% Difference
2	1.28E+00	1.28E+00	0.003%	6.10E-02	6.10E-02	0.002%	6.723E-01	6.723E-01	0.001%
6	1.96E+00	1.96E+00	0.002%	9.31E-02	9.31E-02	0.001%	3.397E+00	3.397E+00	0.001%
10	2.06E+00	2.06E+00	0.002%	9.78E-02	9.78E-02	0.002%	5.942E+00	5.942E+00	0.000%
14	2.07E+00	2.07E+00	0.000%	9.84E-02	9.84E-02	0.002%	7.943E+00	7.943E+00	0.001%
20	2.08E+00	2.08E+00	0.000%	9.86E-02	9.86E-02	0.001%	1.009E+01	1.009E+01	0.002%
28	2.08E+00	2.08E+00	0.002%	9.86E-02	9.86E-02	0.002%	1.187E+01	1.187E+01	0.000%
36	2.08E+00	2.08E+00	0.002%	9.86E-02	9.86E-02	0.002%	1.290E+01	1.290E+01	-0.002%
42	2.08E+00	2.08E+00	0.002%	9.86E-02	9.86E-02	0.002%	1.337E+01	1.337E+01	0.000%
54	2.08E+00	2.08E+00	0.002%	9.86E-02	9.86E-02	0.002%	1.389E+01	1.389E+01	0.004%

Verification Problem 2

Verification Problem 2 considers the same model as Verification Problem 1 but $R(t) = 0$ and $Q_{1,1}(0) = 1.0$ Ci. Radionuclide inventories and pore water concentrations for compartment 1 were calculated using Equations 13 and 5, respectively. The results (Table 6) indicate very little difference between the analytical solution and FOLAT. Differences between the analytical solution and FOLAT increase as the inventory decreases. However, differences are still less than 0.05 percent, and the inventory at those times (42 and 54 years) is nine orders of magnitude smaller than the initial inventory.

Table 6. Results of Verification Problem 2 for Compartment 1

Time (y)	FOLAT Inventory (Ci)	Analytical solution (Ci)	% Difference	FOLAT Conc. (Ci m^{-3})	Analytical solution (Ci m^{-3})	% Difference
2	3.82E-01	3.82E-01	-0.002%	1.81E-02	1.81E-02	0.002%
6	5.55E-02	5.55E-02	-0.005%	2.64E-03	2.64E-03	0.005%
10	8.09E-03	8.09E-03	-0.008%	3.84E-04	3.84E-04	-0.08%
14	1.18E-03	1.18E-03	-0.010%	5.59E-05	5.59E-05	-0.010%
20	6.54E-05	6.54E-05	-0.016%	3.10E-06	3.10E-06	-0.016%
28	1.39E-06	1.39E-06	-0.019%	6.58E-08	6.58E-08	-0.019%
36	2.94E-08	2.94E-08	-0.029%	1.39E-09	1.39E-09	-0.029%
42	1.63E-09	1.63E-09	-0.031%	7.75E-11	7.74E-11	-0.031%
54	5.03E-12	5.03E-12	-0.044%	2.39E-13	2.39E-13	-0.044%

Verification Problem 3

Verification Problem 3 uses the general solution developed by Birchall (1986) to solve a four compartment model with a two member decay chain consisting of Pu-241 ($T1/2$ = 14.4 years) and Am-241 ($T1/2$ = 432 years). Parameters values (Table 7) include the initial inventories of Pu-241 and Am-241 in each of the compartments, compartment-specific partition coefficients, and a constant water flux.

Table 7. Parameters Values for Verification Problem 3

Parameter	Compartment 1	Compartment 2	Compartment 3	Compartment 4
Length (m)	1.0	1.0	1.0	1.0
Width (m)	1.0	1.0	1.0	1.0
Thickness (m)	0.02	0.13	0.15	0.2
Bulk density (g cm^{-3})	1.2	1.5	1.5	1.8
Saturated hydraulic conductivity (m y^{-1})	1710	1710	1710	1710
Porosity	0.2724	0.2724	0.2724	0.2724
Residual moisture content	0.0321	0.0321	0.0321	0.0321
Alpha (m^{-1})	7.71	7.71	7.71	7.71
rn	2.28	2.28	2.28	2.28
Calculated moisture content	0.0659	0.0659	0.0659	0.0659
Water flux, $t = 0$ to $t = \infty$ (m y^{-1})	0.2	0.2	0.2	0.2
Pu-241 partition coefficient (mL g^{-1})	10	15	22	22
Am-241 partition coefficient (mL g^{-1})	50	60	70	70
Pu-241 solubility (mg L^{-1})	infinite	infinite	infinite	infinite
Am-241 solubility (mg L^{-1})	infinite	infinite	infinite	infinite
Pu-241 initial inventory (Ci)	2.40E-05	5.46E-04	6.75E-05	2.72E-11
Am-241 initial inventory (Ci)	4.80E-07	2.93E-06	2.25E-07	2.86E-14

As shown in Figure 2 (for Pu-241) and Figure 3 (for Am-241), there is excellent agreement between the analytical solution and FOLAT. Differences between the analytical expression and FOLAT for the maximum inventory beyond 1-year ranged from 0.0000% for Pu-241 in compartment 3, to 0.1277% for Am-241 in compartment 4.

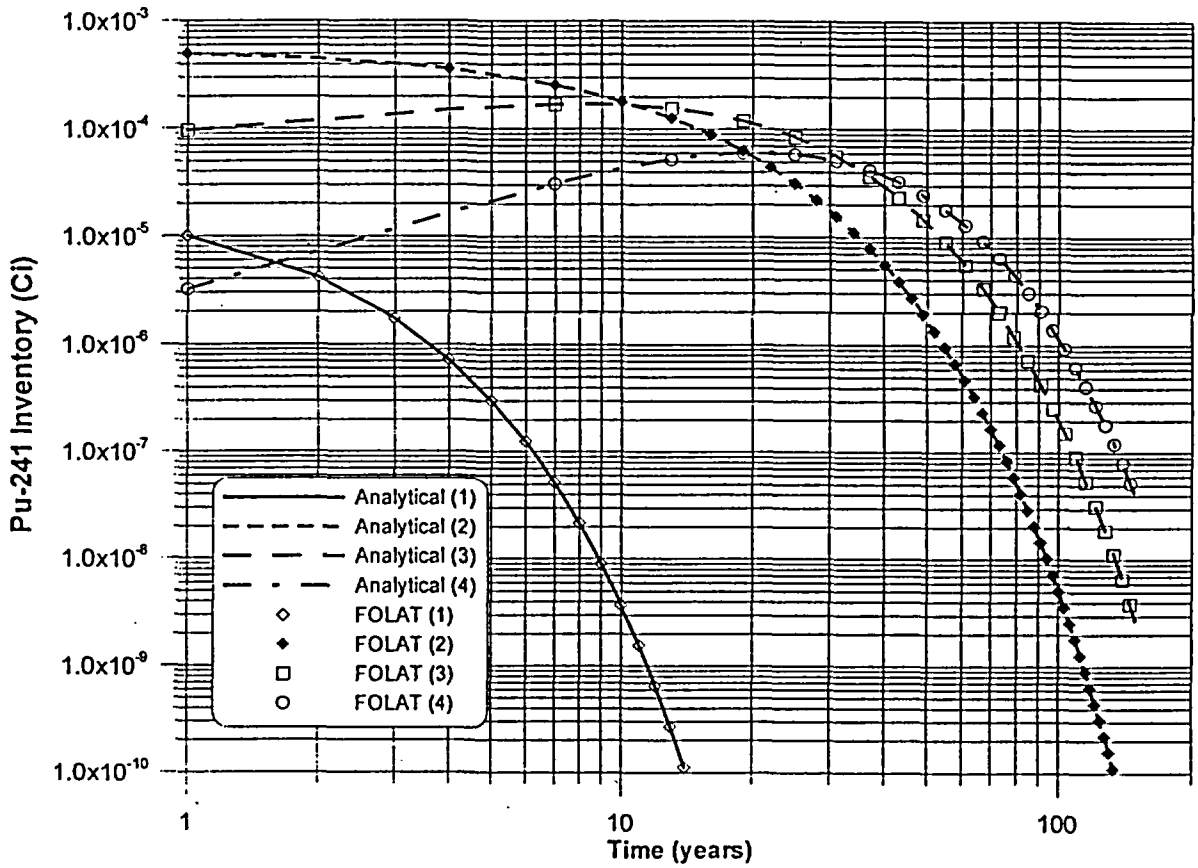

Figure 2. Plutonium-241 inventory as a function of time calculated with the analytical solution developed by Birchall (1986) and FOLAT. The numbers in parentheses in the legend refer to the compartment number.

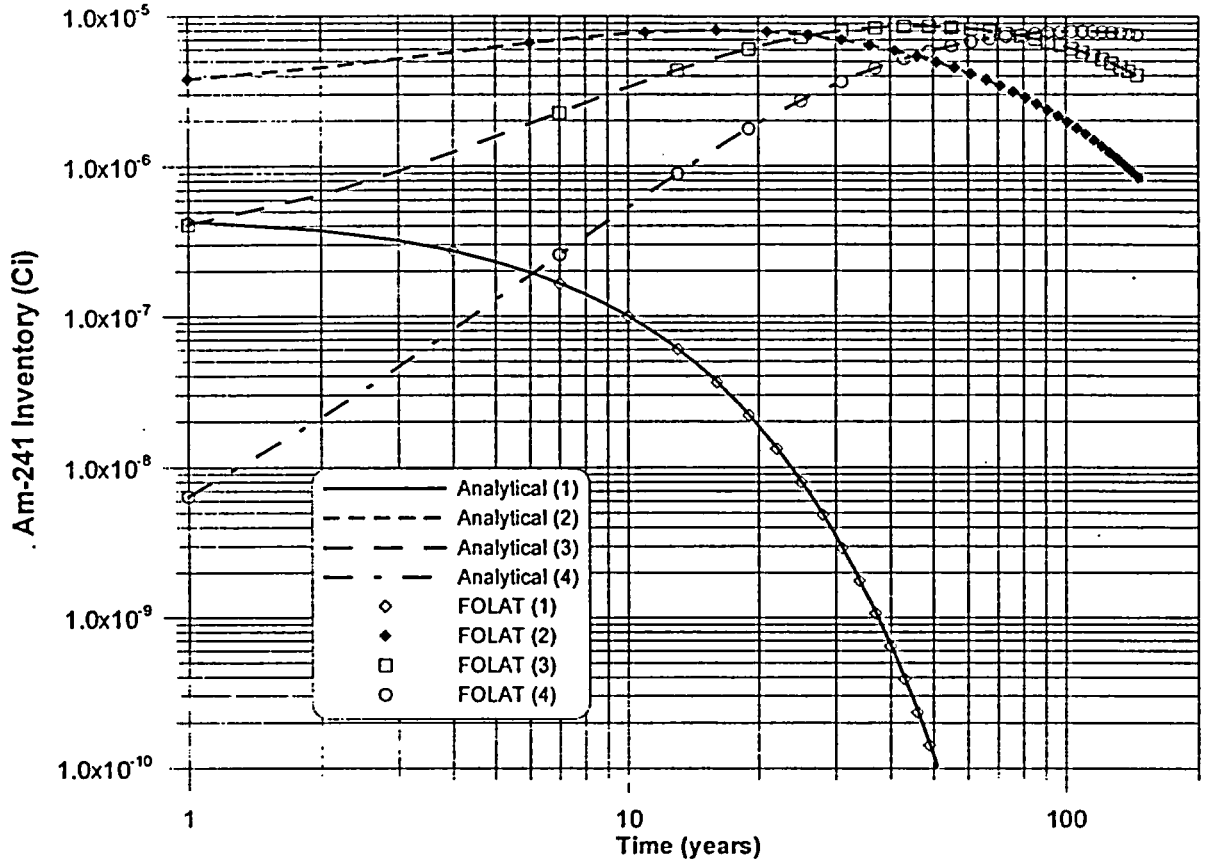

Figure 3. Americium-241 inventory as a function of time calculated with the analytical solution developed by Birchall (1986) and FOLAT. The numbers in parentheses in the legend refer to the compartment number.

Verification Problem 4

Verification Problem 4 checks the solubility limited release function in FOLAT with the analytical solution expressed by Equations 18 and 19. The problem uses the same parameters used in Verification Problem 2, except the initial inventory is set to 1×10^4 Ci. The flux from compartment 1 was also output and compared with output from the GWSCREEN model (Rood 1999), which includes the solubility release model described in Equations 18 and 19. Results (Table 8) show good agreement between FOLAT, the analytical solution, and GWSCREEN. Differences are no greater than 0.163 percent.

Table 8. Inventory and Flux from Compartment 1 for Verification Problem 4

Time (y)	FOLAT Inventory (Ci)	Analytical solution (Ci)	% Difference	FOLAT flux. ($Ci\ y^{-1}$)	GWSCREEN flux ($Ci\ y^{-1}$)	% Difference
2	9.35E+03	9.35E+03	-0.001%	2.59E+02	2.59E+02	0.012%
6	8.07E+03	8.07E+03	-0.002%	2.59E+02	2.59E+02	0.012%
10	6.83E+03	6.83E+03	-0.005%	2.59E+02	2.59E+02	0.012%
14	5.62E+03	5.62E+03	-0.008%	2.59E+02	2.59E+02	0.012%
20	3.86E+03	3.86E+03	-0.015%	2.59E+02	2.59E+02	0.012%
28	1.64E+03	1.64E+03	-0.048%	2.59E+02	2.59E+02	0.012%
36	8.30E+01	8.29E+01	-0.162%	3.94E+01	3.95E+01	0.119%
42	4.61E+00	4.60E+00	-0.162%	2.19E+00	2.19E+00	0.146%
54	1.42E-02	1.42E-02	-0.163%	6.76E-03	6.76E-03	0.115%

CODE BENCHMARKS

The primary output of FOLAT is the radionuclide flux from the unsaturated zone to the aquifer. In this section, we compare the radionuclide flux to the aquifer calculated with FOLAT to radionuclide fluxes calculated using the various solutions to the one-dimensional advection-dispersion equation as implemented in other assessment codes. Because the model formulation for unsaturated transport in FOLAT is quite different from that of the advection-dispersion equation, we do not expect results to match exactly. However, for an assessment model, several values are key for describing the overall behavior system. Namely, the magnitude of the maximum flux, the time of maximum flux, and the cumulative flux. We compare these three values as a means of quantifying the differences between the models. The transport equations are described first followed by four benchmark exercises.

Transport Equation

The governing transport equation in one dimension for a radionuclide with chain decay is given by

$$\theta Rd_j \frac{\partial C_j}{\partial t} = \nabla \cdot \theta D_j \nabla C_j + \nabla \cdot q C_j - \lambda_j \theta Rd_j C_j + \sum_{k=1-N}^{j \neq k} f_{i,j} \lambda_k Rd_k C_k + R_j \qquad (20)$$

where Rd is the retardation coefficient (unitless), D is the dispersion coefficient ($m^2\ time^{-1}$), $f_{i,j}$ is the decay branching ratio, and other terms as defined previously.

Equation 20 was taken from the formulation in DUST (Sullivan 1996). In FOLAT, the branching ratio for all radioactive progeny is assumed to be 1. Analytical solutions exist for Equation 20 for the limiting case of a single decay chain member subjected to a constant water flux through a homogeneous isotropic porous media of infinite extent and constant dispersion

coefficient and retardation factor. In terms of a Cartesian coordinate system with water flow in the positive x direction, Equation 20 is rewritten as

$$\frac{\partial C}{\partial t} + \frac{q}{R\theta}\frac{\partial C}{\partial x} = \frac{D}{R\theta}\frac{\partial^2 C}{\partial x^2} - \lambda C \tag{21}$$

Assuming an instantaneous release of a unit mass at $x = 0$ and $t = 0$ for the initial conditions, $C = 0$ for all x at $t = 0$, and boundary condition $C = 0$ at $x = \pm\infty$, the solution to Equation 21 is

$$C(x,t) = \frac{1}{\theta Rd\sqrt{4\pi Dt/Rd}}\exp\left(\frac{(x-(q/\theta)t/Rd)^2}{4Dt/Rd} - \lambda t\right) \tag{22}$$

The radionuclide flux at a distance x for the initial and boundary conditions given in Equation 22 is (Codell et al. 1982)

$$F(x,t) = \theta\left(\frac{q}{\theta}C(x,t) - D\frac{\partial C}{\partial x}\right) = \frac{x+\frac{qt}{\theta Rd}}{4\sqrt{D\pi t^3/Rd}}\exp\left(\frac{-\left(x-\frac{qt}{\theta Rd}\right)}{4Dt/Rd} - \lambda t\right) \tag{23}$$

Equation 22 is formulated in terms of an instantaneous release. Solutions for an arbitrary release can be arrived at through the use of the convolution integral as given in Equation 23.

$$F = \int_0^t F(x,t-\tau)R(\tau)d\tau \tag{24}$$

Equation 24 in implemented in the GWSCREEN model (Rood 1999) for a source function described by

$$R(t) = \kappa Qo\exp(-(\kappa+\lambda_1)t) \tag{25}$$

Both the GWSCREEN and DUST models are used in benchmark exercises 1 through 4. An implicit finite difference scheme is used to solve Equation 20 in DUST while GWSCREEN uses Equation 24 and 25 to calculate the flux to the aquifer from an initial inventory of a radionuclide in the soil.

Benchmark Exercise 1

Benchmark problem 1 simulates 1 Ci of I-129 in a 10 m surface soil compartment and compares the flux to the aquifer calculated with FOLAT and GWSCREEN for different values of the dispersion coefficient. The dispersion coefficient describes the spreading of the plume as it is transported down the soil column and is given by

$$D = \alpha_L \, q / \theta \tag{26}$$

where α_L = the dispersivity (m). Radionuclide independent transport parameter values are listed in Table 9. For this exercise, I-129 is assumed to have infinite solubility and a partition coefficient of zero throughout the model domain.

Table 9. Parameter Values used in Benchmark Exercises 1, 2, and 3

Parameter	Value
Thickness of source compartment (m)	10
Length of source and unsaturated compartments (m)	382
Width of source and unsaturated compartments (m)	518
Bulk density of source compartment (g cm^{-3})	1.26
Saturated hydraulic conductivity of source and unsaturated zone (m y^{-1})	1.710
Total porosity of source and unsaturated zone	0.272
Residual moisture content of source and unsaturated zone	0.0321
Alpha of source and unsaturated zone (m^{-1})	7.51
n of source and unsaturated zone	2.3
Water flux through source and unsaturated zone (m y^{-1})	0.02
Number of unsaturated compartments in FOLAT	17
Thickness of each unsaturated compartment (m)	5
Total unsaturated thickness (m)	85

The results of Benchmark Exercise 1 (Figure 4) show differences between FOLAT and GWSCREEN depending on the dispersivity value chosen. Using a dispersivity value (α_L) of 2.5 m results in nearly identical maximum flux and time of maximum flux (Table 10). Dispersion effects are represented in FOLAT by the number of compartments the unsaturated zone is composed of. The amount of spreading that occurs at a distance x can be described by the standard deviation of the plume as given by

$$\sigma = \sqrt{2Dt / Rd} = \sqrt{\frac{2\alpha_L q t}{\theta Rd}} \tag{27}$$

Under a state-state water flux, the mean contaminant travel time (tt) at a distance, x from the source can be approximated by

$$tt = \frac{x\theta}{q} Rd \tag{28}$$

Substitution of tt in Equation 28 for t in Equation 27 gives

$$\sigma = \sqrt{2\alpha_L x} \tag{29}$$

which provides a measure of the amount of plume spreading at the time of maximum flux. For an α_L value of 2.5 m and $x = 85$ m, $\sigma = 20.6$ m. Because we are interested in mimicking the behavior of the transport equation (Equations 20 and 23) in FOLAT, a relationship between σ and the thickness of the unsaturated zone compartments can be made and used as a general rule of thumb when selecting the number of compartments to use in a model simulation. The ratio of the two values is 5 m/20.61 m = 0.243. Or in other words, compartment size should be about 25% of the standard deviation of the plume at the time of maximum flux. This rule of thumb is tested in Benchmark Exercise 4.

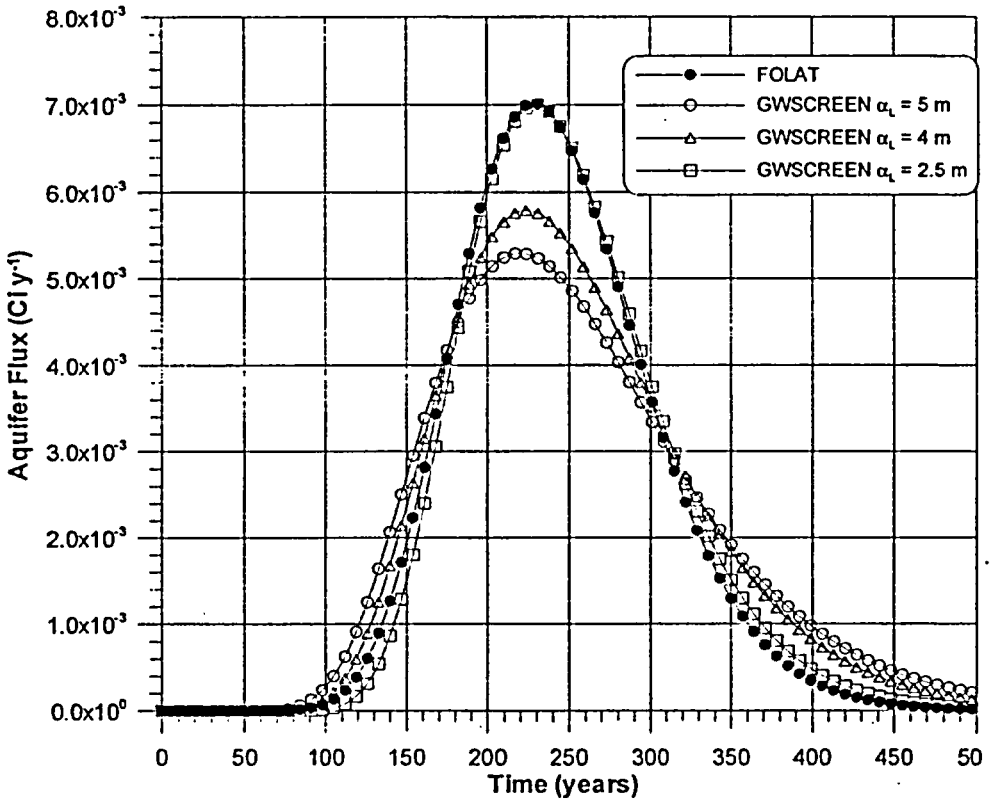

Figure 4. Iodine-129 flux to the aquifer for Benchmark Exercise 1 for GWSCREEN and FOLAT for different values of α_L.

Table 10 Maximum Flux and Time of Maximum Flux in Benchmark 1 for FOLAT and GWSCREEN

	α (m)	GWSCREEN	FOLAT	GWSCREEN/FOLAT Ratio
Maximum flux (Ci y^{-1})		5.29×10^{-3}	7.01×10^{-3}	0.755
Time of maximum flux (years)	5	217	227	0.956
Cumulative flux (Ci)		0.362	0.423	0.857
Maximum flux (Ci y^{-1})		5.79×10^{-3}	7.01×10^{-3}	0.826
Time of maximum flux (years)	4	223	227	0.982
Cumulative flux (Ci)[a]		0.385	0.423	0.910
Maximum flux (Ci y^{-1})		6.50×10^{-3}	7.01×10^{-3}	0.927
Time of maximum flux (years)	3	226	227	0.996
Cumulative flux (Ci)[a]		0.392	0.423	0.926
Maximum flux (Ci y^{-1})		7.00×10^{-3}	7.01×10^{-3}	0.999
Time of maximum flux (years)	2.5	229	227	1.01
Cumulative flux (Ci)[a]		0.406	0.423	0.959

[a] Cumulative flux at time of maximum flux

Benchmark Exercise 2

Benchmark Exercise 2 simulates a 10 Ci U-238 source using the same model parameters as used in Benchmark Exercise 1. The partition coefficient is assumed to be zero in all layers and a solubility limit of 25 mg L^{-1} in the source is imposed. Initial pore water concentrations exceed the solubility of uranium in this case and results in a constant release rate from the source for some portion of the release history. A dispersivity value α_L of 2.5 m was used in the GWSCREEN simulation. Results (Figure 5) shows reasonably good agreement between GWSCREEN and FOLAT. The maximum flux to the aquifer for GWSCREEN and FOLAT was 3.30×10^{-2} Ci y^{-1} and 3.31×10^{-2} Ci y^{-1}, respectively. The time of maximum flux occurred during the period from 380 years to 407 years for GWSCREEN and from 370 to 393 years for FOLAT. Cumulative flux at 500 years was 1.9842 Ci and 1.9832 Ci for GWSCREEN and FOLAT, respectively.

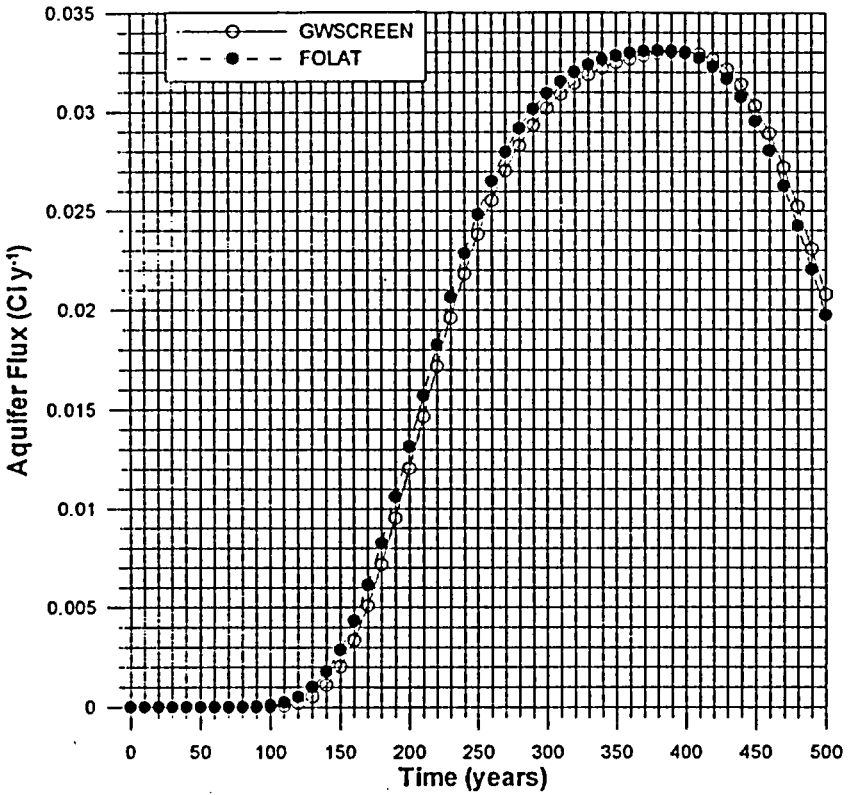

Figure 5. Uranium-238 aquifer flux as a function of time for Benchmark Exercise 2.

Benchmark Exercise 3

Benchmark Exercise 3 simulates a 2 Ci U-238 source using the same model parameters as used in Benchmark Exercise 1. The partition coefficient is assumed to be 0.5 mL g^{-1} in the source and 1.0 mL g^{-1} in the unsaturated zone. A dispersivity value of 2.5 m was used in the GWSCREEN simulation. Results (Figure 6) shows reasonably good agreement between GWSCREEN and FOLAT. The maximum flux to the aquifer for GWSCREEN and FOLAT was 4.72×10^{-4} Ci y^{-1} and 4.66×10^{-4} Ci y^{-1}, respectively. The time of maximum flux occurred during the period from 6880 years to 7004 years for GWSCREEN and from 7000 to 7006 years for FOLAT. The cumulative flux at t = 9000 years was 1.61 and 1.64 Ci for GWSCREEN and FOLAT, respectively.

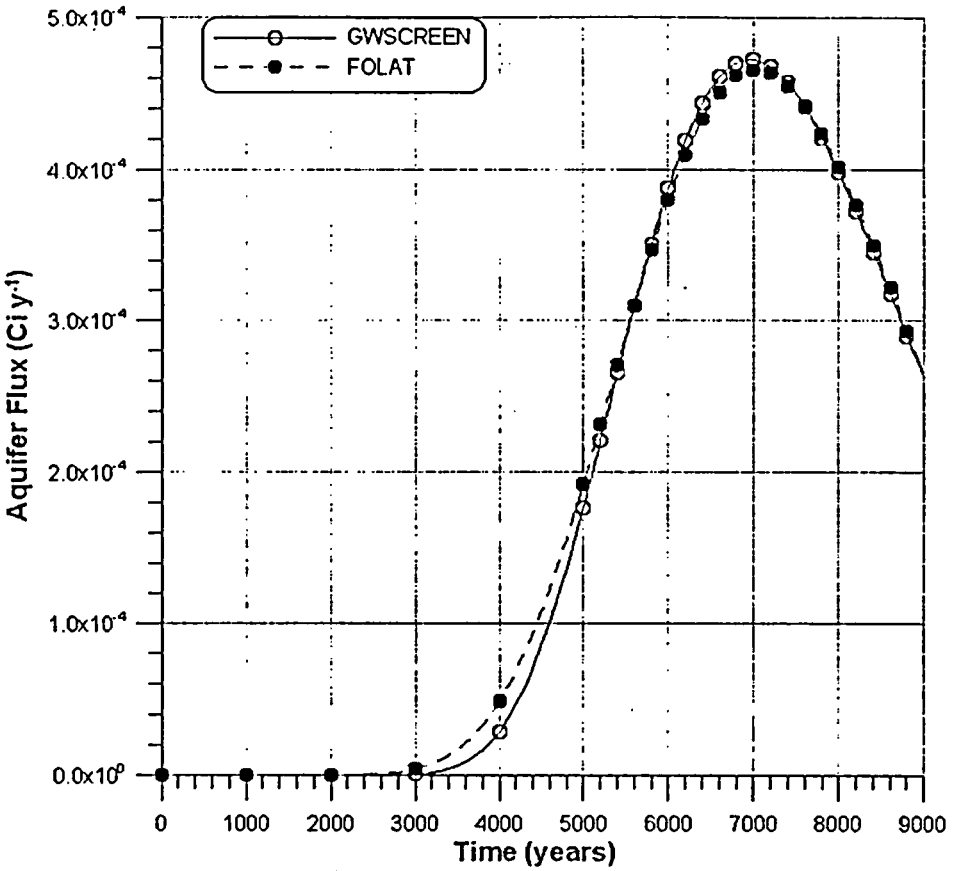

Figure 6. Uranium-238 aquifer flux as a function of time for Benchmark Exercise 3.

Benchmark Exercise 4

Benchmark Exercise 4 considers a 6.42 Ci U-238 source in a 10.6 m surface soil compartment and compares the aquifer flux of U-238 and its progeny, U-234, Th-230, and Ra-226 with values calculated by DUST. Transport parameters (Table 11) include a time-varying water flux of 0.075 m y^{-1} for the first 40 years decreasing to 0.02 m y^{-1} for all future times. The change in water flux simulates the placement of an infiltration reducing barrier over the contaminated soil. Moisture contents in DUST are not allowed to change as a function of time. Therefore, a fixed value of 0.05 was used throughout the model domain.

The number of compartments in the model was selected based on the procedure outlined earlier in Equations 27 through 29. The standard deviation of the plume at the aquifer is

$$\sigma = \sqrt{2 \times 4.1\,\text{m} \times 82.3\,\text{m}} = 27.978\,\text{m} \tag{30}$$

Therefore, the estimated compartment thickness is 0.243 × 27.978 m = 6.31 m. Some minor adjustments were made to this value so that the number of compartments in the unsaturated zone represented a whole number.

Table 11 Parameter Values used in Benchmark Exercise 4

Parameter	Value
Thickness of source compartment (m)	10.6
Finite difference node spacing in DUST (m)	0.5
Length of source and unsaturated compartments (m)	382
Width of source and unsaturated compartments (m)	518
Initial U-238 activity (Ci)[a]	6.42
Bulk density of source compartment (g cm^{-3})	1.26
Moisture content throughout model domain (m^3 m^{-3})	0.05
Uranium partition coefficient, 0 m to 23.26 m (mL g^{-1})	0.2
Thorium partition coefficient, 0 m to 23.26 m (mL g^{-1})	5.0
Radium partition coefficient, 0 m to 23.26 m (mL g^{-1})	0.4
Uranium partition coefficient, 23.27 m to 92.9 m (mL g^{-1})	3
Thorium partition coefficient, 23.27 m to 92.9 m (mL g^{-1})	1000
Radium partition coefficient, 23.27 m to 92.9 m (mL g^{-1})	20
Water flux through source and unsaturated zone, 0 to 40 years (m y^{-1})	0.075
Water flux through source and unsaturated zone, 45 to ∞ years (m y^{-1})	0.02
Dispersivity in Dust (m)	4.1
Number of unsaturated compartments in FOLAT	13
Thickness of each unsaturated compartment (m)	6.33
Total unsaturated thickness (m)	82.3

a. In DUST, this was simulated by setting the initial U-238 pore water concentration at 10 pCi cm^{-3} in the first 22 nodes of the model domain.

Results of the simulation are presented in Table 12 and Figure 7. There are some substantial differences between the aquifer fluxes predicted by both models during the early time periods. In general, FOLAT predicts an earlier arrival of the U-238 plume compared to DUST. However, when the magnitude of the maximum flux and time of maximum flux are compared between the two models, there is no substantial difference, especially when considering the accuracy of a prospective analysis extending out to tens of thousands of years in the future. There appears to be about a 9% mass balance error in the DUST simulation for U-238, which might be improved on with smaller time steps or more finite difference nodes. Nevertheless, these benchmarks illustrate that FOLAT can mimic the behavior of the advection-dispersion transport equation.

CONCLUSIONS

The FOLAT model offers a simple approach to addressing transport of radionuclides in the unsaturated zone. Its formulation incorporates many of the processes that govern the overall behavior of radionuclide transport in the unsaturated zone when performing a prospective analysis. Some of the major processes include temporally and spatially variable water fluxes, spatially variable sorption coefficients, and radioactive progeny that travel according to their own properties. FOLAT approximates the classic advection-dispersion equation solution using ordinary differential equations. Comparison of aquifer fluxes between FOLAT and advection-dispersion show little meaningful difference when the model is properly parameterized. The

dispersion show little meaningful difference when the model is properly parameterized. The model is intended for assessment purposes and is not intended to be a mechanistic model of radionuclide transport in the unsaturated zone.

Table 12 Maximum Flux , Time of Maximum Flux, and Total Activity Released in Benchmark 4 for FOLAT and DUST

	DUST	FOLAT	DUST/FOLAT Ratio
Maximum U-238 flux (Ci y^{-1})	4.79E-04	5.23E-04	0.92
Maximum U-234 flux (Ci y^{-1})	1.88E-05	2.36E-05	0.80
Maximum Th-232 flux (Ci y^{-1})	6.09E-09	7.39E-09	0.82
Maximum Ra-226 flux (Ci y^{-1})	2.96E-07	3.59E-07	0.82
Time of U-238 maximum flux (years)	1.34E+04	1.56E+04	0.86
Time of U-234 maximum flux (years)	1.49E+04	1.67E+04	0.89
Time of Th-232 maximum flux (years)	3.12E+04	2.98E+04	1.05
Time of Ra-226 maximum flux (years)	3.23E+04	3.18E+04	1.02
Cumulative U-238 flux (Ci)	5.84E+00	6.42E+00	0.91
Cumulative U-234 flux (Ci)	2.57E-01	3.037E-01	0.85
Cumulative Th-230 flux (Ci)	6.26E-04	7.364E-04	0.85
Cumulative Ra-226 flux (Ci)	3.05E-02	3.587E-02	0.85

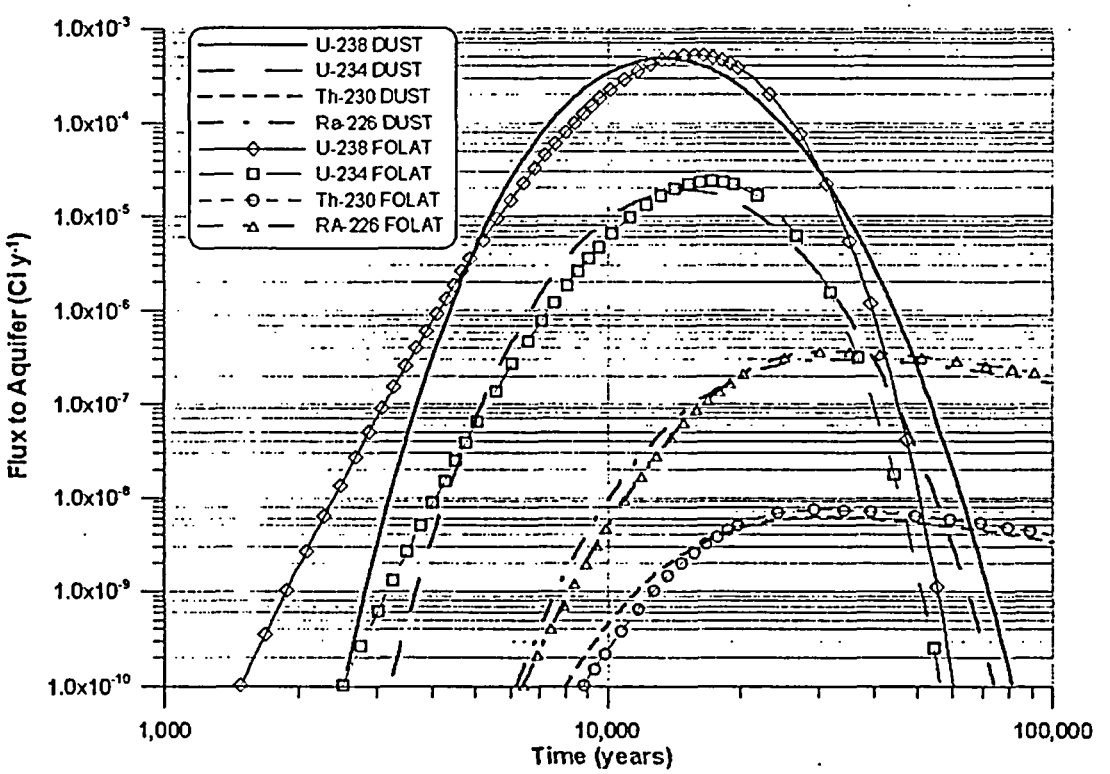

Figure 7. Flux to the aquifer as a function of time as calculated by DUST and FOLAT for the U-238 decay chain.

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APPENDIX A: FOWL: WATER FLUX PRE-PROCESSOR FOR FOLAT

APPENDIX A: A WATER FLUX PREPROCESSOR FOR FOLAT

The FOWL program is a water flux preprocessor that calculates a FOLAT compatible water flux file for either steady state or transient fluxes. The net water flux as a function of time at the surface (first compartment) is provided by the user. The FOWL model then calculates the flux through all reaming compartments as a function of time. A simple water-balance model coupled with the soil-specific moisture characteristic curve is used to calculate the net water flux through each compartment. Unit gradient conditions are assumed to exist.

The water balance equation in the first (uppermost) compartment is given by

$$\frac{d\phi_1}{dt} = q(t) - K_1(\theta_1) \tag{A-1}$$

where

ϕ_1 = the total water contained in a unit area of compartment 1 (m),
$q(t)$ = the net water flux into compartment 1 (m y^{-1}),
$K_1(\theta)$ = hydraulic conductivity as a function of moisture content (m y^{-1}).

The water balance equation in the remainder of the compartments is given by

$$\frac{d\phi_i}{dt} = K_{i-1}(\theta_{i-1}) - K_i(\theta_i) \tag{A-2}$$

where i≠1.

The moisture content for the i^{th} compartment is given by

$$\theta_i = \frac{\phi_i}{T_i} \tag{A-3}$$

where

T_i = the thickness of the i^{th} compartment (m),

The hydraulic conductivity as a function of moisture content is calculated using (van Genuchten 1978)

$$K(\theta) = K_{sat}\left(\frac{\theta - \theta_r}{\theta_s - \theta_r}\right)^{1/2}\left\{1 - \left[1 - \left(\frac{\theta - \theta_r}{\theta_s - \theta_r}\right)^{\frac{1}{m}}\right]^m\right\}^2 \tag{A-4}$$

and

$$\left(\frac{\theta-\theta_r}{\theta_s-\theta_r}\right)=\left(\frac{1}{1+\alpha n}\right)^m \quad \text{(A-5)}$$

where

θ = volumetric moisture content ($m^3\ m^{-3}$),
θ_r = residual moisture content ($m^3\ m^{-3}$),
θ_s = saturated moisture content ($m^3\ m^{-3}$),
K_{sat} = saturated hydraulic conductivity ($m\ y^{-1}$),
α = fitting parameter (m^{-1}),
n = fitting parameter,
m = $1 - 1/n$.

Equations A-1 and A-2 are solved using a forth-order Runga-Kutta solver described in Press et al. (1992). The water flux (q) through a compartment at time t is then given by the hydraulic conductivity function for the calculated moisture content.

$$q_i(t)=K\left(\frac{\phi_i(t)}{T_i}\right) \quad \text{(A-6)}$$

The preprocessor is written in FORTRAN and is run by constructing two ASCII input files; a parameter definition file and a net infiltration file. The parameter definition file defines model options and compartment properties. Moisture contents are initialized based on the first record in the net infiltration file. Tables A-1 and A-2 describe the structure of these two files.

Execution of FOWL is performed on the command line by typing

[*path*] FOWL [*parameter definition file*]

where *path* is path to the FOWL executable. If the parameter definition file is omitted, then the program will look for the default parameter file name, FOWL.PAR. If this file is not found, the program will abort. The program will echo all input data and raw output to the file, FOWL.OUT. The output file name specified in the parameter definition file contains the FOLAT-formatted water flux in each compartment as a function of time. The number of compartments and compartment dimensions in the FOWL simulation must equal that in the FOLAT simulation.

Table A-1. Parameter Definition File for the FOWL Program

Card	Code variable	Type/format	Units	Description
1	Title	CHAR/A80		Title of run
2	Fileout	CHAR/A60		Output file name
3	Fileppt	CHAR/A60		File containing net water flux in the first compartment as a function of time
4	eps	REAL/*		Required accuracy of solution (1×10^{-6})
4	h1	REAL/*	year	Beginning time step (0.0001 year)
4	hmin	REAL/*	year	Minimum time step (1×10^{-30} year)
5	mlayer	INT/*		Number of compartments in the simulation (maximum

Table A-1. Parameter Definition File for the FOWL Program

Card	Code variable	Type/format	Units	Description
				= 50)
5	nmat	INT/*		Number of material types (must be ≤ mlayer)
5	nkt	INT/*		Number of points to describe hydraulic conductivity curve as a function of moisture content
5	qmax	REAL/*	$m\ y^{-1}$	Maximum infiltration rate for hydraulic conductivity curve vs. moisture content curve
NOTE: Card 6a, 6b, and 6c define the compartment properties and are read nmat number of times. Compartments must be defined in ascending order				
6a	h	INT/*		Beginning compartment number to define compartment properties.
6a	j	INT/*		Ending compartment number to define compartment properties.
6b	thick(*i*)	REAL/*	m	Thickness of compartment i
6c	sk(*i*)	REAL/*	$m\ y^{-1}$	Saturated hydraulic conductivity of compartment i
6c	ths(i)	REAL/*	$m^{-3}\ m^{-3}$	Saturated porosity of compartment i
6c	thr(*i*)	REAL/*	$m^{-3}\ m^{-3}$	Residual moisture content of compartment i
6c	alpha(*i*)	REAL/*	m^{-1}	van Genuchten fitting parameter for compartment i
6c	rn(*i*)	REAL/*		van Genuchten fitting parameter for compartment i
7a	ntimes	INT/*		number of output time periods
NOTE: Card 7b is read ntime number of times				
7b	t1	REAL/*	years	Begin time of output
7b	t2	REAL/*	years	End time of output
7b	tp	REAL/*	years	Print step
8	tmax	REAL/*	years	Maximum time of simulation

Table A-2. Description of the Net Infiltration Rate Input File

Line number	Code variable	Description
1	Junk	Column header (discarded)
2 to $n+1$[a]	q(k,1)	Time in years from the start of the simulation for the k^{th} record
2 to $n+1$[a]	q(k,2)	Net infiltration rate ($m\ y^{-1}$)for the k^{th} record

[a] n is the number of time, of infiltration rate records. A minimum of two records is needed to operate the code.

APPENDIX B: FOLAT INPUT AND OUTPUT FILES

Verification Problem 1

Parameter Definition File

```
Verification problem #1 for FOLAT
'verify1.out'
'verify1.pcp'
'verify1.inv'
1.0e-6  .1    1.0e-20                 eps h1 hmin
3   1   1                              mlayer  nprog nmat
'verify1'                              cname
138                                    mw
1.                                      sol
100.                                   thalf
0. 0.  0.                              y(i)
.1  .5   1.0                           kd(i,j)
0.  0.  0.                             kx(i,j)
$ layer 1-3
1 3
1  1.5  10  10                        thick(1),rho(1) len(1) width(1)
1710,0.2724,0.0321,7.51,2.298          sk(1),ths(1),thr(1),alpha(1),rn(1)
$ output times
1                                      ntimes
0. 466. 2.0                             t1,t2,tp
```

Water Flux File

```
Time(y)  ppt (m/y)
0.0      0.1     0.05      0.025
1.0E5    0.1     0.05      0.025
```

Release Rate File

```
Time (y)      Rel (Ci/y)
0.0           1.0
1e5           1.0
```

Output File

Note: Output is truncated after 60 years

```
   ***************************************************
   *                                                 *
   *     This output was produced by the model:      *
   *                                                 *
   *                      FOLAT                      *
   *    The First-Order-Leach-And-Transport model.   *
   *    A general purpose solver for leaching and    *
   *    subsurface transport of radionuclides in     *
   *  surface or buried locations. Version date:     *
   *                    111602                       *
   *                 Arthur S. Rood                  *
   *                   K-Spar Inc                    *
   *             493 N 4154 E Rigby ID 83442         *
   *                   asr@srv.net                   *
   ***************************************************
 Date: 11/16/2002 Time: 20:19:38.990
 Input File:          verify1.par
 Output File:         verify1.out
 Pecipitation File:   verify1.pcp
 Release File:        verify1.inv
 Number of layers:           3
 Number of progeny           1
 Nuclide Names             verify
 Half lifes (y)          1.000E+02
```

```
Solubility (mg/L)          1.000E+00
Molar Weight (g/mol)       1.380E+02
Kd Value for Member 1 for Each Layer (mL/g)
 1.000E-01  5.000E-01  1.000E+00
Kx Value for Member 1 for Each Layer (1/y)
 0.000E+00  0.000E+00  0.000E+00
Initial Activity for Member 1 for Each Layer (Ci)
 0.000E+00  0.000E+00  0.000E+00
Thickness of each layer (m)
 1.000E+00  1.000E+00  1.000E+00
Bulk Density (g/cm**3)
 1.500E+00  1.500E+00  1.500E+00
K-sat (m/y)
 1.710E+03  1.710E+03  1.710E+03
Porosity
 2.724E-01  2.724E-01  2.724E-01
Residual Moisture Content
 3.210E-02  3.210E-02  3.210E-02
Alpha (1/m)
 7.510E+00  7.510E+00  7.510E+00
Van Genuchten n
 2.298E+00  2.298E+00  2.298E+00
Length (m)
 1.000E+01  1.000E+01  1.000E+01
Width (m)
 1.000E+01  1.000E+01  1.000E+01
----------------------
  Calculated Values
----------------------
Moisture Content in each Layer at each Time
 0.000E+00  6.060E-02  5.613E-02  5.235E-02
 1.000E+05  6.060E-02  5.613E-02  5.235E-02
Decay Constants of each Member (1/y)
 6.931E-03
Solubility of Each Member (Ci/m**3)
 2.593E+01
Initial Pore Water Conconcentration for Member 1 for Each Layer (Ci/m**3)
 0.000E+00  0.000E+00  0.000E+00
Initial Leach Rate for Member 1 for Each Layer (1/y)
 4.748E-01  6.203E-02  1.610E-02
Conversion factors from activity (Ci) to mass (atoms)
 1.683E+20
Conversion factors from mass (atoms) to activity (Ci)
 5.940E-21
----------------------
 Start of Calculation
----------------------
C=Concentration (Ci/m**3), I=Inventory (Ci) F=Flux (Ci/y)

ID  Time (y)      Layer  1    Layer  2    Layer  3
C1  0.0000E+00  0.0000E+00  0.0000E+00  0.0000E+00
I1  0.0000E+00  0.0000E+00  0.0000E+00  0.0000E+00
F1  0.0000E+00  0.0000E+00  0.0000E+00  0.0000E+00
C1  2.0000E+00  6.0955E-02  8.3398E-03  1.9270E-04
I1  2.0000E+00  1.2837E+00  6.7230E-01  2.9913E-02
F1  2.0000E+00  6.0955E-01  4.1699E-02  4.8174E-04
C1  4.0000E+00  8.4213E-02  2.4574E-02  1.2061E-03
I1  4.0000E+00  1.7735E+00  1.9810E+00  1.8722E-01
F1  4.0000E+00  8.4213E-01  1.2287E-01  3.0151E-03
C1  6.0000E+00  9.3087E-02  4.2139E-02  3.2574E-03
I1  6.0000E+00  1.9604E+00  3.3969E+00  5.0566E-01
F1  6.0000E+00  9.3087E-01  2.1069E-01  8.1435E-03
C1  8.0000E+00  9.6472E-02  5.8746E-02  6.2981E-03
I1  8.0000E+00  2.0317E+00  4.7357E+00  9.7769E-01
F1  8.0000E+00  9.6472E-01  2.9373E-01  1.5745E-02
C1  1.0000E+01  9.7764E-02  7.3713E-02  1.0197E-02
I1  1.0000E+01  2.0589E+00  5.9422E+00  1.5829E+00
F1  1.0000E+01  9.7764E-01  3.6856E-01  2.5492E-02
C1  1.2000E+01  9.8257E-02  8.6941E-02  1.4807E-02
```

I1	1.2000E+01	2.0693E+00	7.0086E+00	2.2986E+00
F1	1.2000E+01	9.8257E-01	4.3471E-01	3.7018E-02
C1	1.4000E+01	9.8445E-02	9.8538E-02	1.9990E-02
I1	1.4000E+01	2.0733E+00	7.9434E+00	3.1031E+00
F1	1.4000E+01	9.8445E-01	4.9269E-01	4.9975E-02
C1	1.6000E+01	9.8517E-02	1.0867E-01	2.5622E-02
I1	1.6000E+01	2.0748E+00	8.7601E+00	3.9774E+00
F1	1.6000E+01	9.8517E-01	5.4334E-01	6.4055E-02
C1	1.8000E+01	9.8544E-02	1.1750E-01	3.1597E-02
I1	1.8000E+01	2.0754E+00	9.4723E+00	4.9049E+00
F1	1.8000E+01	9.8544E-01	5.8752E-01	7.8991E-02
C1	2.0000E+01	9.8555E-02	1.2521E-01	3.7821E-02
I1	2.0000E+01	2.0756E+00	1.0093E+01	5.8712E+00
F1	2.0000E+01	9.8555E-01	6.2603E-01	9.4554E-02
C1	2.2000E+01	9.8559E-02	1.3192E-01	4.4219E-02
I1	2.2000E+01	2.0757E+00	1.0634E+01	6.8643E+00
F1	2.2000E+01	9.8559E-01	6.5959E-01	1.1055E-01
C1	2.4000E+01	9.8560E-02	1.3776E-01	5.0723E-02
I1	2.4000E+01	2.0757E+00	1.1106E+01	7.8739E+00
F1	2.4000E+01	9.8560E-01	6.8882E-01	1.2681E-01
C1	2.6000E+01	9.8561E-02	1.4286E-01	5.7277E-02
I1	2.6000E+01	2.0757E+00	1.1516E+01	8.8914E+00
F1	2.6000E+01	9.8561E-01	7.1429E-01	1.4319E-01
C1	2.8000E+01	9.8561E-02	1.4730E-01	6.3836E-02
I1	2.8000E+01	2.0757E+00	1.1874E+01	9.9096E+00
F1	2.8000E+01	9.8561E-01	7.3648E-01	1.5959E-01
C1	3.0000E+01	9.8561E-02	1.5116E-01	7.0361E-02
I1	3.0000E+01	2.0757E+00	1.2186E+01	1.0922E+01
F1	3.0000E+01	9.8561E-01	7.5581E-01	1.7590E-01
C1	3.2000E+01	9.8561E-02	1.5453E-01	7.6818E-02
I1	3.2000E+01	2.0757E+00	1.2457E+01	1.1925E+01
F1	3.2000E+01	9.8561E-01	7.7265E-01	1.9205E-01
C1	3.4000E+01	9.8561E-02	1.5746E-01	8.3184E-02
I1	3.4000E+01	2.0757E+00	1.2694E+01	1.2913E+01
F1	3.4000E+01	9.8561E-01	7.8732E-01	2.0796E-01
C1	3.6000E+01	9.8561E-02	1.6002E-01	8.9435E-02
I1	3.6000E+01	2.0757E+00	1.2900E+01	1.3883E+01
F1	3.6000E+01	9.8561E-01	8.0011E-01	2.2359E-01
C1	3.8000E+01	9.8561E-02	1.6225E-01	9.5554E-02
I1	3.8000E+01	2.0757E+00	1.3079E+01	1.4833E+01
F1	3.8000E+01	9.8561E-01	8.1124E-01	2.3889E-01
C1	4.0000E+01	9.8561E-02	1.6419E-01	1.0153E-01
I1	4.0000E+01	2.0757E+00	1.3236E+01	1.5761E+01
F1	4.0000E+01	9.8561E-01	8.2094E-01	2.5382E-01
C1	4.2000E+01	9.8561E-02	1.6588E-01	1.0735E-01
I1	4.2000E+01	2.0757E+00	1.3372E+01	1.6664E+01
F1	4.2000E+01	9.8561E-01	8.2939E-01	2.6837E-01
C1	4.4000E+01	9.8561E-02	1.6735E-01	1.1301E-01
I1	4.4000E+01	2.0757E+00	1.3491E+01	1.7543E+01
F1	4.4000E+01	9.8561E-01	8.3675E-01	2.8252E-01
C1	4.6000E+01	9.8561E-02	1.6863E-01	1.1850E-01
I1	4.6000E+01	2.0757E+00	1.3594E+01	1.8395E+01
F1	4.6000E+01	9.8561E-01	8.4317E-01	2.9624E-01
C1	4.8000E+01	9.8561E-02	1.6975E-01	1.2381E-01
I1	4.8000E+01	2.0757E+00	1.3684E+01	1.9220E+01
F1	4.8000E+01	9.8561E-01	8.4875E-01	3.0953E-01
C1	5.0000E+01	9.8561E-02	1.7072E-01	1.2896E-01
I1	5.0000E+01	2.0757E+00	1.3763E+01	2.0019E+01
F1	5.0000E+01	9.8561E-01	8.5362E-01	3.2239E-01
C1	5.2000E+01	9.8561E-02	1.7157E-01	1.3393E-01
I1	5.2000E+01	2.0757E+00	1.3831E+01	2.0790E+01
F1	5.2000E+01	9.8561E-01	8.5786E-01	3.3482E-01
C1	5.4000E+01	9.8561E-02	1.7231E-01	1.3872E-01
I1	5.4000E+01	2.0757E+00	1.3890E+01	2.1534E+01
F1	5.4000E+01	9.8561E-01	8.6156E-01	3.4680E-01
C1	5.6000E+01	9.8561E-02	1.7296E-01	1.4334E-01
I1	5.6000E+01	2.0757E+00	1.3942E+01	2.2252E+01
F1	5.6000E+01	9.8561E-01	8.6478E-01	3.5836E-01
C1	5.8000E+01	9.8561E-02	1.7352E-01	1.4780E-01
I1	5.8000E+01	2.0757E+00	1.3988E+01	2.2943E+01

```
 F1  5.8000E+01  9.8561E-01  8.6758E-01  3.6949E-01
```

Verification Problem 2

Parameter Definition File

```
Verification problem #2 for FOLAT
'verify2.out'
'verify2.pcp'
'verify2.inv'
1.0e-6  .1    1.0e-20              eps hl hmin
3   1  1                            mlayer  nprog  nmat
'verify2'                          rel file name
138                                 mw
1.                                   sol
100.                                thalf
1.  0   0                           initial y(i)
.1 .5  1.0                               kd(i,j)
0. 0. 0.                            kx(i.j)
1 3
$ layer 1-3
1  1.5  10  10                      thick(1),rho(1) len(1) width(1)
1710,0.2724,0.0321,7.51,2.298       sk(1),ths(1),thr(1),alpha(1),rn(1)
$ output times
1                                   ntimes
0. 466. 2.0                           t1,t2,tp
```

Water Flux File

```
time     layer1  layer2    layer3   (m/y)
0.0      0.1     0.05      0.025
1.0E5    0.1     0.05      0.025
```

Release File

```
Time (y)       Rel (Ci/y)
0.0            0.0
1e5            0.0
```

Output File

Note: Output is truncated after 60 years

```
   ************************************************
   *                                              *
   *     This output was produced by the model:   *
   *                                              *
   *                      FOLAT                   *
   *    The First-Order-Leach-And-Transport model.*
   *    A general purpose solver for leaching and *
   *    subsurface transport of radionuclides in  *
   *  surface or buried locations. Version date:  *
   *                    111602                    *
   *                 Arthur S. Rood               *
   *                   K-Spar Inc                 *
   *             493 N 4154 E Rigby ID 83442      *
   *                   asr@srv.net                *
   ************************************************
 Date: 11/16/2002 Time: 20:19:42.720
 Input File:          verify2.par
 Output File:         verify2.out
 Pecipitation File:   verify2.pcp
```

```
Release File:        verify2.inv
Number of layers:           3
Number of progeny           1
Nuclide Names             verify
Half lifes (y)             1.000E+02
Solubility (mg/L)          1.000E+00
Molar Weight (g/mol)       1.380E+02
Kd Value for Member 1 for Each Layer (mL/g)
 1.000E-01  5.000E-01  1.000E+00
Kx Value for Member 1 for Each Layer (1/y)
 0.000E+00  0.000E+00  0.000E+00
Initial Activity for Member 1 for Each Layer (Ci)
 1.000E+00  0.000E+00  0.000E+00
Thickness of each layer (m)
 1.000E+00  1.000E+00  1.000E+00
Bulk Density (g/cm**3)
 1.500E+00  1.500E+00  1.500E+00
K-sat (m/y)
 1.710E+03  1.710E+03  1.710E+03
Porosity
 2.724E-01  2.724E-01  2.724E-01
Residual Moisture Content
 3.210E-02  3.210E-02  3.210E-02
Alpha (1/m)
 7.510E+00  7.510E+00  7.510E+00
Van Genuchten n
 2.298E+00  2.298E+00  2.298E+00
Length (m)
 1.000E+01  1.000E+01  1.000E+01
Width (m)
 1.000E+01  1.000E+01  1.000E+01
---------------------
  Calculated Values
---------------------
Moisture Content in each Layer at each Time
 0.000E+00  6.060E-02  5.613E-02  5.235E-02
 1.000E+05  6.060E-02  5.613E-02  5.235E-02
Decay Constants of each Member (1/y)
 6.931E-03
Solubility of Each Member (Ci/m**3)
 2.593E+01
Initial Pore Water Conconcentration for Member 1 for Each Layer (Ci/m**3)
 4.748E-02  0.000E+00  0.000E+00
Initial Leach Rate for Member 1 for Each Layer (1/y)
 4.748E-01  6.203E-02  1.610E-02
Conversion factors from activity (Ci) to mass (atoms)
 1.683E+20
Conversion factors from mass (atoms) to activity (Ci)
 5.940E-21
----------------------
 Start of Calculation
----------------------
C=Concentration (Ci/m**3), I=Inventory (Ci) F=Flux (Ci/y)

ID  Time (y)      Layer  1    Layer  2    Layer  3
C1  0.0000E+00  4.7483E-02  0.0000E+00  0.0000E+00
I1  0.0000E+00  1.0000E+00  0.0000E+00  0.0000E+00
F1  0.0000E+00  4.7483E-01  0.0000E+00  0.0000E+00
C1  2.0000E+00  1.8117E-02  6.9864E-03  2.6418E-04
I1  2.0000E+00  3.8155E-01  5.6319E-01  4.1010E-02
F1  2.0000E+00  1.8117E-01  3.4932E-02  6.6045E-04
C1  4.0000E+00  6.9125E-03  8.7520E-03  7.6373E-04
I1  4.0000E+00  1.4558E-01  7.0553E-01  1.1856E-01
F1  4.0000E+00  6.9125E-02  4.3760E-02  1.9093E-03
C1  6.0000E+00  2.6375E-03  8.6416E-03  1.2822E-03
I1  6.0000E+00  5.5546E-02  6.9663E-01  1.9905E-01
F1  6.0000E+00  2.6375E-02  4.3208E-02  3.2056E-03
C1  8.0000E+00  1.0063E-03  7.9164E-03  1.7471E-03
I1  8.0000E+00  2.1194E-02  6.3817E-01  2.7121E-01
F1  8.0000E+00  1.0063E-02  3.9582E-02  4.3677E-03
```

C1	1.0000E+01	3.8396E-04	7.0447E-03	2.1393E-03
I1	1.0000E+01	8.0864E-03	5.6789E-01	3.3210E-01
F1	1.0000E+01	3.8396E-03	3.5223E-02	5.3483E-03
C1	1.2000E+01	1.4650E-04	6.1936E-03	2.4592E-03
I1	1.2000E+01	3.0854E-03	4.9929E-01	3.8176E-01
F1	1.2000E+01	1.4650E-03	3.0968E-02	6.1480E-03
C1	1.4000E+01	5.5897E-05	5.4173E-03	2.7133E-03
I1	1.4000E+01	1.1772E-03	4.3670E-01	4.2121E-01
F1	1.4000E+01	5.5897E-04	2.7086E-02	6.7834E-03
C1	1.6000E+01	2.1328E-05	4.7276E-03	2.9099E-03
I1	1.6000E+01	4.4917E-04	3.8111E-01	4.5172E-01
F1	1.6000E+01	2.1328E-04	2.3638E-02	7.2747E-03
C1	1.8000E+01	8.1375E-06	4.1217E-03	3.0569E-03
I1	1.8000E+01	1.7138E-04	3.3226E-01	4.7453E-01
F1	1.8000E+01	8.1375E-05	2.0609E-02	7.6422E-03
C1	2.0000E+01	3.1049E-06	3.5919E-03	3.1615E-03
I1	2.0000E+01	6.5390E-05	2.8956E-01	4.9078E-01
F1	2.0000E+01	3.1049E-05	1.7960E-02	7.9039E-03
C1	2.2000E+01	1.1847E-06	3.1297E-03	3.2303E-03
I1	2.2000E+01	2.4949E-05	2.5229E-01	5.0146E-01
F1	2.2000E+01	1.1847E-05	1.5648E-02	8.0758E-03
C1	2.4000E+01	4.5201E-07	2.7267E-03	3.2688E-03
I1	2.4000E+01	9.5195E-06	2.1980E-01	5.0744E-01
F1	2.4000E+01	4.5201E-06	1.3633E-02	8.1721E-03
C1	2.6000E+01	1.7246E-07	2.3755E-03	3.2819E-03
I1	2.6000E+01	3.6321E-06	1.9149E-01	5.0947E-01
F1	2.6000E+01	1.7246E-06	1.1877E-02	8.2048E-03
C1	2.8000E+01	6.5803E-08	2.0695E-03	3.2738E-03
I1	2.8000E+01	1.3858E-06	1.6683E-01	5.0820E-01
F1	2.8000E+01	6.5803E-07	1.0347E-02	8.1844E-03
C1	3.0000E+01	2.5107E-08	1.8029E-03	3.2480E-03
I1	3.0000E+01	5.2877E-07	1.4533E-01	5.0420E-01
F1	3.0000E+01	2.5107E-07	9.0144E-03	8.1200E-03
C1	3.2000E+01	9.5797E-09	1.5706E-03	3.2077E-03
I1	3.2000E+01	2.0175E-07	1.2661E-01	4.9795E-01
F1	3.2000E+01	9.5797E-08	7.8531E-03	8.0193E-03
C1	3.4000E+01	3.6551E-09	1.3683E-03	3.1556E-03
I1	3.4000E+01	7.6978E-08	1.1030E-01	4.8986E-01
F1	3.4000E+01	3.6551E-08	6.8414E-03	7.8890E-03
C1	3.6000E+01	1.3946E-09	1.1920E-03	3.0939E-03
I1	3.6000E+01	2.9371E-08	9.6092E-02	4.8029E-01
F1	3.6000E+01	1.3946E-08	5.9601E-03	7.7349E-03
C1	3.8000E+01	5.3211E-10	1.0385E-03	3.0247E-03
I1	3.8000E+01	1.1206E-08	8.3713E-02	4.6954E-01
F1	3.8000E+01	5.3211E-09	5.1923E-03	7.5617E-03
C1	4.0000E+01	2.0303E-10	9.0468E-04	2.9495E-03
I1	4.0000E+01	4.2758E-09	7.2928E-02	4.5787E-01
F1	4.0000E+01	2.0303E-09	4.5234E-03	7.3739E-03
C1	4.2000E+01	7.7465E-11	7.8813E-04	2.8699E-03
I1	4.2000E+01	1.6314E-09	6.3533E-02	4.4551E-01
F1	4.2000E+01	7.7465E-10	3.9407E-03	7.1748E-03
C1	4.4000E+01	2.9557E-11	6.8660E-04	2.7870E-03
I1	4.4000E+01	6.2248E-10	5.5349E-02	4.3264E-01
F1	4.4000E+01	2.9557E-10	3.4330E-03	6.9675E-03
C1	4.6000E+01	1.1277E-11	5.9815E-04	2.7019E-03
I1	4.6000E+01	2.3751E-10	4.8218E-02	4.1943E-01
F1	4.6000E+01	1.1277E-10	2.9907E-03	6.7547E-03
C1	4.8000E+01	4.3029E-12	5.2109E-04	2.6154E-03
I1	4.8000E+01	9.0620E-11	4.2007E-02	4.0600E-01
F1	4.8000E+01	4.3029E-11	2.6055E-03	6.5384E-03
C1	5.0000E+01	1.6418E-12	4.5396E-04	2.5282E-03
I1	5.0000E+01	3.4576E-11	3.6595E-02	3.9247E-01
F1	5.0000E+01	1.6418E-11	2.2698E-03	6.3206E-03
C1	5.2000E+01	6.2641E-13	3.9548E-04	2.4411E-03
I1	5.2000E+01	1.3192E-11	3.1881E-02	3.7894E-01
F1	5.2000E+01	6.2641E-12	1.9774E-03	6.1027E-03
C1	5.4000E+01	2.3901E-13	3.4453E-04	2.3544E-03
I1	5.4000E+01	5.0336E-12	2.7774E-02	3.6549E-01
F1	5.4000E+01	2.3901E-12	1.7227E-03	5.8860E-03

```
C1  5.6000E+01  9.1193E-14  3.0015E-04  2.2687E-03
I1  5.6000E+01  1.9206E-12  2.4196E-02  3.5217E-01
F1  5.6000E+01  9.1193E-13  1.5007E-03  5.6716E-03
C1  5.8000E+01  3.4795E-14  2.6148E-04  2.1841E-03
I1  5.8000E+01  7.3279E-13  2.1079E-02  3.3906E-01
F1  5.8000E+01  3.4795E-13  1.3074E-03  5.4604E-03
C1  6.0000E+01  1.3276E-14  2.2780E-04  2.1012E-03
I1  6.0000E+01  2.7960E-13  1.8363E-02  3.2617E-01
F1  6.0000E+01  1.3276E-13  1.1390E-03  5.2529E-03
```

Verification Problem 3

Parameter Definition File

```
Verification problem #3 for FOLAT - Comparison with BOXRAD for Lab 2 Problem
'verify3.out'
'verify3.pcp'
'verify3.inv'
1.0e-4  .001    1.0e-30                eps h1 hmin
4   2    4                            mlayer  nprog  nmat
'Pu-241' 'Am-241'                      rel file name
241   241                              mw
10000.  10000                                 sol
14.4   432                             thalf
$
2.40E-05  5.46E-04  6.75e-5 2.72e-11     initial pu241 layer 1-4
4.80E-07  2.93E-06  2.25e-7 2.86e-14   initial am241 layer 1-4
10  15  22  22                         kd pu241 layer 1-4
50  60  70  70                         kd am241 layer 1-4
0.  0.  0.  0.                         kx pu241 layer 1-4
0.  0.  0.  0.                         kx am241 layer 1-4
$ layer 1
1 1
0.02  1.2  1  1                        thick(1),rho(1) len(1) width(1)
1710,0.2724,0.0321,7.51,2.298        sk(1),ths(1),thr(1),alpha(1),rn(1)
$  layer 2
2  2
0.13  1.5  1  1                        thick(2),rho(2)  len(2) width(2)
1710,0.2724,0.0321,7.51,2.298        sk(2),ths(2),thr(2),alpha(2),rn(2)
$  layer 3
3  3
0.15  1.5   1  1                       thick(3),rho(3)  len(2) width(2)
1710,0.2724,0.0321,7.51,2.298        sk(3),ths(3),thr(3),alpha(3),rn(3)
$  layer 4
4  4
0.2  1.8   1  1                        thick(3),rho(3)  len(2) width(2)
1710,0.2724,0.0321,7.51,2.298        sk(3),ths(3),thr(3),alpha(3),rn(3)
$
1                                    ntimes
0.00   150    1.00E+00          tstart tend tp
```

Water Flux File

```
time      layer1  layer2    layer3  layer 4 (m/y)
0.0       0.2     0.2        0.2    0.2
1.0E5     0.2     0.2        0.2    0.2
```

Release File

```
Time (y)       Rel (Ci/y)
0.0            0.0
1e5            0.0
```

Output File

Note: Output is truncated after 100 years

```
**************************************************
*                                                *
*     This output was produced by the model:     *
*                                                *
*                      FOLAT                     *
*    The First-Order-Leach-And-Transport model.   *
*    A general purpose solver for leaching and    *
*    subsurface transport of radionuclides in     *
*  surface or buried locations. Version date:     *
*                      111602                    *
*                  Arthur S. Rood                *
*                    K-Spar Inc                  *
*              493 N 4154 E Rigby ID 83442       *
*                    asr@srv.net                 *
**************************************************
Date: 11/16/2002 Time: 20:19:45.520
Input File:          verify3.par
Output File:         verify3.out
Pecipitation File:   verify3.pcp
Release File:        verify3.inv
Number of layers:           4
Number of progeny           2
Nuclide Names             Pu-241 Am-241
Half lifes (y)          1.440E+01  4.320E+02
Solubility (mg/L)       1.000E+04  1.000E+04
Molar Weight (g/mol)    2.410E+02  2.410E+02
Kd Value for Member 1 for Each Layer (mL/g)
 1.000E+01  1.500E+01  2.200E+01  2.200E+01
Kd Value for Member 2 for Each Layer (mL/g)
 5.000E+01  6.000E+01  7.000E+01  7.000E+01
Kx Value for Member 1 for Each Layer (1/y)
 0.000E+00  0.000E+00  0.000E+00  0.000E+00
Kx Value for Member 2 for Each Layer (1/y)
 0.000E+00  0.000E+00  0.000E+00  0.000E+00
Initial Activity for Member 1 for Each Layer (Ci)
 2.400E-05  5.460E-04  6.750E-05  2.720E-11
Initial Activity for Member 2 for Each Layer (Ci)
 4.800E-07  2.930E-06  2.250E-07  2.860E-14
Thickness of each layer (m)
 2.000E-02  1.300E-01  1.500E-01  2.000E-01
Bulk Density (g/cm**3)
 1.200E+00  1.500E+00  1.500E+00  1.800E+00
K-sat (m/y)
 1.710E+03  1.710E+03  1.710E+03  1.710E+03
Porosity
 2.724E-01  2.724E-01  2.724E-01  2.724E-01
Residual Moisture Content
 3.210E-02  3.210E-02  3.210E-02  3.210E-02
Alpha (1/m)
 7.510E+00  7.510E+00  7.510E+00  7.510E+00
Van Genuchten n
 2.298E+00  2.298E+00  2.298E+00  2.298E+00
Length (m)
 1.000E+00  1.000E+00  1.000E+00  1.000E+00
Width (m)
 1.000E+00  1.000E+00  1.000E+00  1.000E+00
---------------------
  Calculated Values
---------------------
Moisture Content in each Layer at each Time
 0.000E+00  6.591E-02  6.591E-02  6.591E-02  6.591E-02
 1.000E+05  6.591E-02  6.591E-02  6.591E-02  6.591E-02
Decay Constants of each Member (1/y)
 4.814E-02  1.605E-03
Solubility of Each Member (Ci/m**3)
```

```
 1.031E+06  3.437E+04
Initial Pore Water Conconcentration for Member 1 for Each Layer (Ci/m**3)
 9.945E-05  1.861E-04  1.361E-05  3.429E-12
Initial Pore Water Conconcentration for Member 2 for Each Layer (Ci/m**3)
 1.989E-06  9.988E-07  4.536E-08  3.605E-15
Initial Leach Rate for Member 1 for Each Layer (1/y)
 8.288E-01  6.818E-02  4.032E-02  2.521E-02
Initial Leach Rate for Member 2 for Each Layer (1/y)
 1.665E-01  1.708E-02  1.269E-02  7.932E-03
Conversion factors from activity (Ci) to mass (atoms)
 2.424E+19  7.272E+20
Conversion factors from mass (atoms) to activity (Ci)
 4.125E-20  1.375E-21
----------------------
 Start of Calculation
----------------------
C=Concentration (Ci/m**3), I=Inventory (Ci) F=Flux (Ci/y)

ID  Time (y)      Layer 1    Layer 2    Layer 3    Layer 4
C1  0.0000E+00  9.9454E-05  1.8612E-04  1.3609E-05  3.4286E-12
I1  0.0000E+00  2.4000E-05  5.4600E-04  6.7500E-05  2.7200E-11
F1  0.0000E+00  1.9891E-05  3.7224E-05  2.7218E-06  6.8573E-13
C2  0.0000E+00  1.9891E-06  9.9878E-07  4.5364E-08  3.6051E-15
I2  0.0000E+00  4.8000E-07  2.9300E-06  2.2500E-07  2.8600E-14
F2  0.0000E+00  7.9912E-08  5.0049E-08  2.8554E-09  2.2687E-16
C1  1.0000E+00  4.1379E-05  1.6991E-04  1.9329E-05  4.0354E-07
I1  1.0000E+00  9.9855E-06  4.9845E-04  9.5869E-05  3.2013E-06
F1  1.0000E+00  8.2758E-06  3.3982E-05  3.8658E-06  8.0708E-08
C2  1.0000E+00  3.5715E-07  3.2299E-07  2.6005E-08  2.5343E-10
I2  1.0000E+00  4.2905E-07  3.7817E-06  4.0983E-07  6.3897E-09
F2  1.0000E+00  7.1430E-08  6.4597E-08  5.2009E-09  5.0685E-11
C1  2.0000E+00  1.7216E-05  1.5301E-04  2.3918E-05  9.0351E-07
I1  2.0000E+00  4.1546E-06  4.4888E-04  1.1863E-04  7.1677E-06
F1  2.0000E+00  3.4433E-06  3.0603E-05  4.7835E-06  1.8070E-07
C2  2.0000E+00  3.0997E-07  3.8693E-07  4.1008E-08  8.3898E-10
I2  2.0000E+00  3.7237E-07  4.5304E-06  6.4628E-07  2.1153E-08
F2  2.0000E+00  6.1994E-08  7.7387E-08  8.2016E-09  1.6780E-10
C1  3.0000E+00  7.1631E-06  1.3694E-04  2.7479E-05  1.4668E-06
I1  3.0000E+00  1.7286E-06  4.0173E-04  1.3629E-04  1.1636E-05
F1  3.0000E+00  1.4326E-06  2.7389E-05  5.4959E-06  2.9335E-07
C2  3.0000E+00  2.6537E-07  4.4232E-07  5.8587E-08  1.8164E-09
I2  3.0000E+00  3.1879E-07  5.1790E-06  9.2332E-07  4.5798E-08
F2  3.0000E+00  5.3074E-08  8.8465E-08  1.1717E-08  3.6329E-10
C1  4.0000E+00  2.9803E-06  1.2221E-04  3.0145E-05  2.0655E-06
I1  4.0000E+00  7.1920E-07  3.5852E-04  1.4951E-04  1.6386E-05
F1  4.0000E+00  5.9606E-07  2.4442E-05  6.0289E-06  4.1310E-07
C2  4.0000E+00  2.2571E-07  4.8983E-07  7.8118E-08  3.2245E-09
I2  4.0000E+00  2.7115E-07  5.7352E-06  1.2311E-06  8.1300E-08
F2  4.0000E+00  4.5142E-08  9.7967E-08  1.5624E-08  6.4490E-10
C1  5.0000E+00  1.2400E-06  1.0892E-04  3.2044E-05  2.6770E-06
I1  5.0000E+00  2.9923E-07  3.1952E-04  1.5894E-04  2.1237E-05
F1  5.0000E+00  2.4800E-07  2.1784E-05  6.4088E-06  5.3539E-07
C2  5.0000E+00  1.9137E-07  5.3026E-07  9.9064E-08  5.0842E-09
I2  5.0000E+00  2.2990E-07  6.2085E-06  1.5612E-06  1.2819E-07
F2  5.0000E+00  3.8274E-08  1.0605E-07  1.9813E-08  1.0168E-09
C1  6.0000E+00  5.1592E-07  9.7012E-05  3.3297E-05  3.2832E-06
I1  6.0000E+00  1.2450E-07  2.8459E-04  1.6515E-04  2.6046E-05
F1  6.0000E+00  1.0318E-07  1.9402E-05  6.6595E-06  6.5664E-07
C2  6.0000E+00  1.6200E-07  5.6438E-07  1.2097E-07  7.4023E-09
I2  6.0000E+00  1.9462E-07  6.6081E-06  1.9064E-06  1.8663E-07
F2  6.0000E+00  3.2400E-08  1.1288E-07  2.4193E-08  1.4805E-09
C1  7.0000E+00  2.1465E-07  8.6382E-05  3.4010E-05  3.8702E-06
I1  7.0000E+00  5.1800E-08  2.5341E-04  1.6869E-04  3.0703E-05
F1  7.0000E+00  4.2931E-08  1.7276E-05  6.8020E-06  7.7404E-07
C2  7.0000E+00  1.3704E-07  5.9293E-07  1.4344E-07  1.0174E-08
I2  7.0000E+00  1.6463E-07  6.9424E-06  2.2606E-06  2.5652E-07
F2  7.0000E+00  2.7408E-08  1.1859E-07  2.8688E-08  2.0348E-09
C1  8.0000E+00  8.9309E-08  7.6906E-05  3.4275E-05  4.4273E-06
I1  8.0000E+00  2.1552E-08  2.2561E-04  1.7000E-04  3.5122E-05
F1  8.0000E+00  1.7862E-08  1.5381E-05  6.8551E-06  8.8546E-07
```

C2	8.0000E+00	1.1588E-07	6.1656E-07	1.6617E-07	1.3386E-08
I2	8.0000E+00	1.3921E-07	7.2191E-06	2.6187E-06	3.3750E-07
F2	8.0000E+00	2.3175E-08	1.2331E-07	3.3233E-08	2.6772E-09
C1	9.0000E+00	3.7158E-08	6.8466E-05	3.4173E-05	4.9468E-06
I1	9.0000E+00	8.9670E-09	2.0085E-04	1.6950E-04	3.9244E-05
F1	9.0000E+00	7.4317E-09	1.3693E-05	6.8346E-06	9.8936E-07
C2	9.0000E+00	9.7966E-08	6.3586E-07	1.8887E-07	1.7017E-08
I2	9.0000E+00	1.1769E-07	7.4451E-06	2.9766E-06	4.2906E-07
F2	9.0000E+00	1.9593E-08	1.2717E-07	3.7775E-08	3.4035E-09
C1	1.0000E+01	1.5460E-08	6.0949E-05	3.3772E-05	5.4232E-06
I1	1.0000E+01	3.7308E-09	1.7880E-04	1.6751E-04	4.3024E-05
F1	1.0000E+01	3.0921E-09	1.2190E-05	6.7545E-06	1.0846E-06
C2	1.0000E+01	8.2816E-08	6.5136E-07	2.1135E-07	2.1043E-08
I2	1.0000E+01	9.9488E-08	7.6264E-06	3.3309E-06	5.3057E-07
F2	1.0000E+01	1.6563E-08	1.3027E-07	4.2270E-08	4.2087E-09
C1	1.1000E+01	6.4324E-09	5.4258E-05	3.3132E-05	5.8532E-06
I1	1.1000E+01	1.5523E-09	1.5917E-04	1.6433E-04	4.6434E-05
F1	1.1000E+01	1.2865E-09	1.0852E-05	6.6264E-06	1.1706E-06
C2	1.1000E+01	7.0006E-08	6.6350E-07	2.3342E-07	2.5435E-08
I2	1.1000E+01	8.4099E-08	7.7686E-06	3.6787E-06	6.4129E-07
F2	1.1000E+01	1.4001E-08	1.3270E-07	4.6684E-08	5.0869E-09
C1	1.2000E+01	2.6763E-09	4.8300E-05	3.2302E-05	6.2348E-06
I1	1.2000E+01	6.4584E-10	1.4169E-04	1.6021E-04	4.9461E-05
F1	1.2000E+01	5.3526E-10	9.6601E-06	6.4604E-06	1.2470E-06
C2	1.2000E+01	5.9175E-08	6.7270E-07	2.5494E-07	3.0161E-08
I2	1.2000E+01	7.1089E-08	7.8764E-06	4.0178E-06	7.6044E-07
F2	1.2000E+01	1.1835E-08	1.3454E-07	5.0987E-08	6.0321E-09
C1	1.3000E+01	1.1135E-09	4.2997E-05	3.1325E-05	6.5673E-06
I1	1.3000E+01	2.6871E-10	1.2613E-04	1.5537E-04	5.2099E-05
F1	1.3000E+01	2.2270E-10	8.5994E-06	6.2651E-06	1.3135E-06
C2	1.3000E+01	5.0020E-08	6.7933E-07	2.7579E-07	3.5188E-08
I2	1.3000E+01	6.0090E-08	7.9540E-06	4.3465E-06	8.8721E-07
F2	1.3000E+01	1.0004E-08	1.3587E-07	5.5159E-08	7.0376E-09
C1	1.4000E+01	4.6329E-10	3.8276E-05	3.0239E-05	6.8511E-06
I1	1.4000E+01	1.1180E-10	1.1228E-04	1.4998E-04	5.4351E-05
F1	1.4000E+01	9.2658E-11	7.6552E-06	6.0477E-06	1.3702E-06
C2	1.4000E+01	4.2281E-08	6.8369E-07	2.9591E-07	4.0485E-08
I2	1.4000E+01	5.0793E-08	8.0051E-06	4.6634E-06	1.0207E-06
F2	1.4000E+01	8.4563E-09	1.3674E-07	5.9181E-08	8.0969E-09
C1	1.5000E+01	1.9276E-10	3.4073E-05	2.9072E-05	7.0874E-06
I1	1.5000E+01	4.6516E-11	9.9956E-05	1.4419E-04	5.6226E-05
F1	1.5000E+01	3.8552E-11	6.8146E-06	5.8144E-06	1.4175E-06
C2	1.5000E+01	3.5740E-08	6.8608E-07	3.1521E-07	4.6017E-08
I2	1.5000E+01	4.2935E-08	8.0331E-06	4.9676E-06	1.1602E-06
F2	1.5000E+01	7.1479E-09	1.3722E-07	6.3042E-08	9.2034E-09
C1	1.6000E+01	8.0200E-11	3.0332E-05	2.7851E-05	7.2780E-06
I1	1.6000E+01	1.9354E-11	8.8981E-05	1.3814E-04	5.7737E-05
F1	1.6000E+01	1.6040E-11	6.0664E-06	5.5702E-06	1.4556E-06
C2	1.6000E+01	3.0210E-08	6.8675E-07	3.3366E-07	5.1753E-08
I2	1.6000E+01	3.6292E-08	8.0408E-06	5.2584E-06	1.3049E-06
F2	1.6000E+01	6.0420E-09	1.3735E-07	6.6731E-08	1.0351E-08
C1	1.7000E+01	3.3368E-11	2.7001E-05	2.6597E-05	7.4249E-06
I1	1.7000E+01	8.0524E-12	7.9210E-05	1.3192E-04	5.8903E-05
F1	1.7000E+01	6.6736E-12	5.4003E-06	5.3194E-06	1.4850E-06
C2	1.7000E+01	2.5536E-08	6.8591E-07	3.5122E-07	5.7662E-08
I2	1.7000E+01	3.0677E-08	8.0310E-06	5.5352E-06	1.4538E-06
F2	1.7000E+01	5.1072E-09	1.3718E-07	7.0244E-08	1.1532E-08
C1	1.8000E+01	1.3883E-11	2.4037E-05	2.5328E-05	7.5307E-06
I1	1.8000E+01	3.3503E-12	7.0513E-05	1.2563E-04	5.9743E-05
F1	1.8000E+01	2.7767E-12	4.8073E-06	5.0657E-06	1.5061E-06
C2	1.8000E+01	2.1585E-08	6.8376E-07	3.6788E-07	6.3713E-08
I2	1.8000E+01	2.5930E-08	8.0058E-06	5.7978E-06	1.6064E-06
F2	1.8000E+01	4.3170E-09	1.3675E-07	7.3576E-08	1.2743E-08
C1	1.9000E+01	5.7763E-12	2.1397E-05	2.4059E-05	7.5982E-06
I1	1.9000E+01	1.3939E-12	6.2770E-05	1.1933E-04	6.0278E-05
F1	1.9000E+01	1.1553E-12	4.2795E-06	4.8118E-06	1.5196E-06
C2	1.9000E+01	1.8245E-08	6.8047E-07	3.8364E-07	6.9877E-08
I2	1.9000E+01	2.1918E-08	7.9673E-06	6.0460E-06	1.7618E-06
F2	1.9000E+01	3.6491E-09	1.3609E-07	7.6727E-08	1.3975E-08

C1	2.0000E+01	2.4033E-12	1.9048E-05	2.2801E-05	7.6302E-06
I1	2.0000E+01	5.7997E-13	5.5878E-05	1.1309E-04	6.0532E-05
F1	2.0000E+01	4.8066E-13	3.8096E-06	4.5602E-06	1.5260E-06
C2	2.0000E+01	1.5422E-08	6.7620E-07	3.9848E-07	7.6129E-08
I2	2.0000E+01	1.8527E-08	7.9174E-06	6.2801E-06	1.9195E-06
F2	2.0000E+01	3.0845E-09	1.3524E-07	7.9697E-08	1.5226E-08
C1	2.1000E+01	9.9993E-13	1.6956E-05	2.1564E-05	7.6296E-06
I1	2.1000E+01	2.4130E-13	4.9743E-05	1.0696E-04	6.0527E-05
F1	2.1000E+01	1.9999E-13	3.3913E-06	4.3128E-06	1.5259E-06
C2	2.1000E+01	1.3036E-08	6.7108E-07	4.1244E-07	8.2443E-08
I2	2.1000E+01	1.5661E-08	7.8574E-06	6.5000E-06	2.0787E-06
F2	2.1000E+01	2.6072E-09	1.3422E-07	8.2487E-08	1.6489E-08
C1	2.2000E+01	4.1604E-13	1.5094E-05	2.0356E-05	7.5992E-06
I1	2.2000E+01	1.0040E-13	4.4281E-05	1.0096E-04	6.0286E-05
F1	2.2000E+01	8.3207E-14	3.0189E-06	4.0711E-06	1.5198E-06
C2	2.2000E+01	1.1019E-08	6.6524E-07	4.2551E-07	8.8796E-08
I2	2.2000E+01	1.3238E-08	7.7890E-06	6.7060E-06	2.2388E-06
F2	2.2000E+01	2.2039E-09	1.3305E-07	8.5102E-08	1.7759E-08
C1	2.3000E+01	1.7310E-13	1.3437E-05	1.9182E-05	7.5421E-06
I1	2.3000E+01	4.1772E-14	3.9419E-05	9.5140E-05	5.9833E-05
F1	2.3000E+01	3.4619E-14	2.6874E-06	3.8364E-06	1.5084E-06
C2	2.3000E+01	9.3144E-09	6.5877E-07	4.3772E-07	9.5166E-08
I2	2.3000E+01	1.1190E-08	7.7132E-06	6.8984E-06	2.3994E-06
F2	2.3000E+01	1.8629E-09	1.3175E-07	8.7544E-08	1.9033E-08
C1	2.4000E+01	7.2019E-14	1.1962E-05	1.8047E-05	7.4610E-06
I1	2.4000E+01	1.7380E-14	3.5090E-05	8.9511E-05	5.9189E-05
F1	2.4000E+01	1.4404E-14	2.3923E-06	3.6094E-06	1.4922E-06
C2	2.4000E+01	7.8732E-09	6.5177E-07	4.4910E-07	1.0153E-07
I2	2.4000E+01	9.4583E-09	7.6313E-06	7.0777E-06	2.5600E-06
F2	2.4000E+01	1.5746E-09	1.3035E-07	8.9819E-08	2.0307E-08
C1	2.5000E+01	2.9965E-14	1.0648E-05	1.6955E-05	7.3585E-06
I1	2.5000E+01	7.2310E-15	3.1237E-05	8.4093E-05	5.8376E-05
F1	2.5000E+01	5.9929E-15	2.1296E-06	3.3909E-06	1.4717E-06
C2	2.5000E+01	6.6551E-09	6.4432E-07	4.5966E-07	1.0788E-07
I2	2.5000E+01	7.9949E-09	7.5441E-06	7.2442E-06	2.7200E-06
F2	2.5000E+01	1.3310E-09	1.2886E-07	9.1932E-08	2.1576E-08
C1	2.6000E+01	1.2467E-14	9.4790E-06	1.5907E-05	7.2372E-06
I1	2.6000E+01	3.0086E-15	2.7807E-05	7.8896E-05	5.7414E-05
F1	2.6000E+01	2.4934E-15	1.8958E-06	3.1814E-06	1.4474E-06
C2	2.6000E+01	5.6254E-09	6.3650E-07	4.6944E-07	1.1419E-07
I2	2.6000E+01	6.7579E-09	7.4525E-06	7.3983E-06	2.8792E-06
F2	2.6000E+01	1.1251E-09	1.2730E-07	9.3888E-08	2.2839E-08
C1	2.7000E+01	5.1871E-15	8.4382E-06	1.4905E-05	7.0997E-06
I1	2.7000E+01	1.2518E-15	2.4754E-05	7.3928E-05	5.6323E-05
F1	2.7000E+01	1.0374E-15	1.6876E-06	2.9811E-06	1.4199E-06
C2	2.7000E+01	4.7550E-09	6.2837E-07	4.7847E-07	1.2045E-07
I2	2.7000E+01	5.7123E-09	7.3573E-06	7.5406E-06	3.0370E-06
F2	2.7000E+01	9.5101E-10	1.2567E-07	9.5694E-08	2.4091E-08
C1	2.8000E+01	2.1582E-15	7.5117E-06	1.3951E-05	6.9481E-06
I1	2.8000E+01	5.2081E-16	2.2036E-05	6.9193E-05	5.5120E-05
F1	2.8000E+01	4.3164E-16	1.5023E-06	2.7901E-06	1.3896E-06
C2	2.8000E+01	4.0193E-09	6.1998E-07	4.8677E-07	1.2665E-07
I2	2.8000E+01	4.8285E-09	7.2591E-06	7.6715E-06	3.1933E-06
F2	2.8000E+01	8.0387E-10	1.2400E-07	9.7355E-08	2.5330E-08
C1	2.9000E+01	8.9794E-16	6.6869E-06	1.3043E-05	6.7845E-06
I1	2.9000E+01	2.1669E-16	1.9616E-05	6.4691E-05	5.3823E-05
F1	2.9000E+01	1.7959E-16	1.3374E-06	2.6086E-06	1.3569E-06
C2	2.9000E+01	3.3975E-09	6.1139E-07	4.9439E-07	1.3277E-07
I2	2.9000E+01	4.0814E-09	7.1585E-06	7.7915E-06	3.3476E-06
F2	2.9000E+01	6.7949E-10	1.2228E-07	9.8877E-08	2.6555E-08
C1	3.0000E+01	3.7360E-16	5.9526E-06	1.2182E-05	6.6111E-06
I1	3.0000E+01	9.0157E-17	1.7463E-05	6.0422E-05	5.2447E-05
F1	3.0000E+01	7.4720E-17	1.1905E-06	2.4364E-06	1.3222E-06
C2	3.0000E+01	2.8718E-09	6.0264E-07	5.0134E-07	1.3881E-07
I2	3.0000E+01	3.4499E-09	7.0560E-06	7.9010E-06	3.4998E-06
F2	3.0000E+01	5.7436E-10	1.2053E-07	1.0027E-07	2.7762E-08
C1	3.1000E+01	1.5544E-16	5.2990E-06	1.1367E-05	6.4296E-06
I1	3.1000E+01	3.7511E-17	1.5545E-05	5.6381E-05	5.1007E-05
F1	3.1000E+01	3.1088E-17	1.0598E-06	2.2735E-06	1.2859E-06
C2	3.1000E+01	2.4275E-09	5.9377E-07	5.0766E-07	1.4475E-07

I2	3.1000E+01	2.9162E-09	6.9521E-06	8.0007E-06	3.6496E-06
F2	3.1000E+01	4.8549E-10	1.1875E-07	1.0153E-07	2.8950E-08
C1	3.2000E+01	6.4673E-17	4.7172E-06	1.0598E-05	6.2416E-06
I1	3.2000E+01	1.5607E-17	1.3838E-05	5.2565E-05	4.9516E-05
F1	3.2000E+01	1.2935E-17	9.4344E-07	2.1196E-06	1.2483E-06
C2	3.2000E+01	2.0519E-09	5.8480E-07	5.1339E-07	1.5059E-07
I2	3.2000E+01	2.4650E-09	6.8472E-06	8.0909E-06	3.7969E-06
F2	3.2000E+01	4.1038E-10	1.1696E-07	1.0268E-07	3.0118E-08
C1	3.3000E+01	2.6908E-17	4.1992E-06	9.8725E-06	6.0488E-06
I1	3.3000E+01	6.4935E-18	1.2319E-05	4.8966E-05	4.7986E-05
F1	3.3000E+01	5.3817E-18	8.3985E-07	1.9745E-06	1.2098E-06
C2	3.3000E+01	1.7344E-09	5.7578E-07	5.1854E-07	1.5632E-07
I2	3.3000E+01	2.0836E-09	6.7416E-06	8.1721E-06	3.9414E-06
F2	3.3000E+01	3.4688E-10	1.1516E-07	1.0371E-07	3.1265E-08
C1	3.4000E+01	1.1196E-17	3.7381E-06	9.1896E-06	5.8525E-06
I1	3.4000E+01	2.7017E-18	1.0966E-05	4.5579E-05	4.6429E-05
F1	3.4000E+01	2.2391E-18	7.4763E-07	1.8379E-06	1.1705E-06
C2	3.4000E+01	1.4661E-09	5.6673E-07	5.2315E-07	1.6194E-07
I2	3.4000E+01	1.7612E-09	6.6356E-06	8.2449E-06	4.0831E-06
F2	3.4000E+01	2.9321E-10	1.1335E-07	1.0463E-07	3.2389E-08
C1	3.5000E+01	4.6581E-18	3.3277E-06	8.5476E-06	5.6541E-06
I1	3.5000E+01	1.1241E-18	9.7620E-06	4.2395E-05	4.4855E-05
F1	3.5000E+01	9.3161E-19	6.6554E-07	1.7095E-06	1.1308E-06
C2	3.5000E+01	1.2392E-09	5.5767E-07	5.2726E-07	1.6744E-07
I2	3.5000E+01	1.4887E-09	6.5295E-06	8.3095E-06	4.2218E-06
F2	3.5000E+01	2.4785E-10	1.1153E-07	1.0545E-07	3.3489E-08
C1	3.6000E+01	1.9380E-18	2.9623E-06	7.9451E-06	5.4545E-06
I1	3.6000E+01	4.6769E-19	8.6901E-06	3.9407E-05	4.3271E-05
F1	3.6000E+01	3.8761E-19	5.9246E-07	1.5890E-06	1.0909E-06
C2	3.6000E+01	1.0475E-09	5.4861E-07	5.3088E-07	1.7282E-07
I2	3.6000E+01	1.2584E-09	6.4235E-06	8.3665E-06	4.3574E-06
F2	3.6000E+01	2.0950E-10	1.0972E-07	1.0618E-07	3.4564E-08
C1	3.7000E+01	8.0635E-19	2.6370E-06	7.3803E-06	5.2548E-06
I1	3.7000E+01	1.9459E-19	7.7359E-06	3.6606E-05	4.1688E-05
F1	3.7000E+01	1.6127E-19	5.2741E-07	1.4761E-06	1.0510E-06
C2	3.7000E+01	8.8543E-10	5.3959E-07	5.3404E-07	1.7808E-07
I2	3.7000E+01	1.0637E-09	6.3178E-06	8.4164E-06	4.4898E-06
F2	3.7000E+01	1.7709E-10	1.0792E-07	1.0681E-07	3.5615E-08
C1	3.8000E+01	3.3549E-19	2.3475E-06	6.8515E-06	5.0560E-06
I1	3.8000E+01	8.0961E-20	6.8865E-06	3.3983E-05	4.0111E-05
F1	3.8000E+01	6.7099E-20	4.6950E-07	1.3703E-06	1.0112E-06
C2	3.8000E+01	7.4844E-10	5.3060E-07	5.3676E-07	1.8320E-07
I2	3.8000E+01	8.9911E-10	6.2126E-06	8.4594E-06	4.6191E-06
F2	3.8000E+01	1.4969E-10	1.0612E-07	1.0735E-07	3.6640E-08
C1	3.9000E+01	1.3959E-19	2.0897E-06	6.3569E-06	4.8589E-06
I1	3.9000E+01	3.3685E-20	6.1303E-06	3.1530E-05	3.8546E-05
F1	3.9000E+01	2.7917E-20	4.1794E-07	1.2714E-06	9.7178E-07
C2	3.9000E+01	6.3264E-10	5.2167E-07	5.3909E-07	1.8820E-07
I2	3.9000E+01	7.6000E-10	6.1080E-06	8.4959E-06	4.7452E-06
F2	3.9000E+01	1.2653E-10	1.0433E-07	1.0782E-07	3.7640E-08
C1	4.0000E+01	5.8077E-20	1.8603E-06	5.8948E-06	4.6640E-06
I1	4.0000E+01	1.4015E-20	5.4572E-06	2.9238E-05	3.7001E-05
F1	4.0000E+01	1.1615E-20	3.7205E-07	1.1790E-06	9.3281E-07
C2	4.0000E+01	5.3476E-10	5.1281E-07	5.4102E-07	1.9307E-07
I2	4.0000E+01	6.4241E-10	6.0043E-06	8.5264E-06	4.8679E-06
F2	4.0000E+01	1.0695E-10	1.0256E-07	1.0820E-07	3.8614E-08
C1	4.1000E+01	2.4164E-20	1.6560E-06	5.4635E-06	4.4721E-06
I1	4.1000E+01	5.8311E-21	4.8580E-06	2.7098E-05	3.5478E-05
F1	4.1000E+01	4.8327E-21	3.3120E-07	1.0927E-06	8.9442E-07
C2	4.1000E+01	4.5202E-10	5.0403E-07	5.4259E-07	1.9781E-07
I2	4.1000E+01	5.4302E-10	5.9014E-06	8.5512E-06	4.9874E-06
F2	4.1000E+01	9.0404E-11	1.0081E-07	1.0852E-07	3.9562E-08
C1	4.2000E+01	1.0054E-20	1.4742E-06	5.0612E-06	4.2836E-06
I1	4.2000E+01	2.4261E-21	4.3246E-06	2.5103E-05	3.3983E-05
F1	4.2000E+01	2.0107E-21	2.9483E-07	1.0122E-06	8.5673E-07
C2	4.2000E+01	3.8208E-10	4.9532E-07	5.4383E-07	2.0242E-07
I2	4.2000E+01	4.5900E-10	5.7995E-06	8.5706E-06	5.1037E-06
F2	4.2000E+01	7.6416E-11	9.9065E-08	1.0877E-07	4.0484E-08
C1	4.3000E+01	4.1829E-21	1.3123E-06	4.6864E-06	4.0991E-06

I1	4.3000E+01	1.0094E-21	3.8497E-06	2.3244E-05	3.2519E-05
F1	4.3000E+01	8.3659E-22	2.6246E-07	9.3728E-07	8.1981E-07
C2	4.3000E+01	3.2297E-10	4.8671E-07	5.4474E-07	2.0690E-07
I2	4.3000E+01	3.8798E-10	5.6987E-06	8.5850E-06	5.2167E-06
F2	4.3000E+01	6.4593E-11	9.7343E-08	1.0895E-07	4.1381E-08
C1	4.4000E+01	1.7404E-21	1.1682E-06	4.3374E-06	3.9188E-06
I1	4.4000E+01	4.1998E-22	3.4270E-06	2.1513E-05	3.1088E-05
F1	4.4000E+01	3.4807E-22	2.3364E-07	8.6749E-07	7.8375E-07
C2	4.4000E+01	2.7300E-10	4.7820E-07	5.4534E-07	2.1126E-07
I2	4.4000E+01	3.2796E-10	5.5991E-06	8.5946E-06	5.3264E-06
F2	4.4000E+01	5.4599E-11	9.5640E-08	1.0907E-07	4.2251E-08
C1	4.5000E+01	7.2410E-22	1.0399E-06	4.0128E-06	3.7430E-06
I1	4.5000E+01	1.7474E-22	3.0507E-06	1.9903E-05	2.9694E-05
F1	4.5000E+01	1.4482E-22	2.0799E-07	8.0255E-07	7.4860E-07
C2	4.5000E+01	2.3076E-10	4.6979E-07	5.4567E-07	2.1548E-07
I2	4.5000E+01	2.7721E-10	5.5006E-06	8.5997E-06	5.4330E-06
F2	4.5000E+01	4.6152E-11	9.3959E-08	1.0913E-07	4.3097E-08
C1	4.6000E+01	3.0127E-22	9.2575E-07	3.7109E-06	3.5721E-06
I1	4.6000E+01	7.2703E-23	2.7158E-06	1.8406E-05	2.8338E-05
F1	4.6000E+01	6.0255E-23	1.8515E-07	7.4218E-07	7.1442E-07
C2	4.6000E+01	1.9505E-10	4.6149E-07	5.4572E-07	2.1958E-07
I2	4.6000E+01	2.3432E-10	5.4034E-06	8.6006E-06	5.5364E-06
F2	4.6000E+01	3.9011E-11	9.2298E-08	1.0914E-07	4.3917E-08
C1	4.7000E+01	1.2535E-22	8.2410E-07	3.4304E-06	3.4062E-06
I1	4.7000E+01	3.0249E-23	2.4176E-06	1.7015E-05	2.7022E-05
F1	4.7000E+01	2.5070E-23	1.6482E-07	6.8609E-07	6.8124E-07
C2	4.7000E+01	1.6488E-10	4.5330E-07	5.4553E-07	2.2356E-07
I2	4.7000E+01	1.9807E-10	5.3075E-06	8.5975E-06	5.6366E-06
F2	4.7000E+01	3.2975E-11	9.0660E-08	1.0911E-07	4.4712E-08
C1	4.8000E+01	5.2153E-23	7.3361E-07	3.1700E-06	3.2454E-06
I1	4.8000E+01	1.2585E-23	2.1521E-06	1.5723E-05	2.5747E-05
F1	4.8000E+01	1.0431E-23	1.4672E-07	6.3400E-07	6.4909E-07
C2	4.8000E+01	1.3937E-10	4.4522E-07	5.4510E-07	2.2741E-07
I2	4.8000E+01	1.6742E-10	5.2129E-06	8.5908E-06	5.7338E-06
F2	4.8000E+01	2.7873E-11	8.9044E-08	1.0902E-07	4.5483E-08
C1	4.9000E+01	2.1699E-23	6.5306E-07	2.9283E-06	3.0900E-06
I1	4.9000E+01	5.2363E-24	1.9158E-06	1.4524E-05	2.4513E-05
F1	4.9000E+01	4.3398E-24	1.3061E-07	5.8567E-07	6.1799E-07
C2	4.9000E+01	1.1780E-10	4.3725E-07	5.4445E-07	2.3115E-07
I2	4.9000E+01	1.4152E-10	5.1196E-06	8.5805E-06	5.8280E-06
F2	4.9000E+01	2.3561E-11	8.7451E-08	1.0889E-07	4.6229E-08
C1	5.0000E+01	9.0281E-24	5.8135E-07	2.7042E-06	2.9398E-06
I1	5.0000E+01	2.1787E-24	1.7054E-06	1.3413E-05	2.3322E-05
F1	5.0000E+01	1.8056E-24	1.1627E-07	5.4084E-07	5.8797E-07
C2	5.0000E+01	9.9577E-11	4.2941E-07	5.4360E-07	2.3476E-07
I2	5.0000E+01	1.1962E-10	5.0277E-06	8.5670E-06	5.9191E-06
F2	5.0000E+01	1.9915E-11	8.5881E-08	1.0872E-07	4.6953E-08
C1	5.1000E+01	3.7563E-24	5.1752E-07	2.4964E-06	2.7951E-06
I1	5.1000E+01	9.0646E-25	1.5182E-06	1.2382E-05	2.2174E-05
F1	5.1000E+01	7.5126E-25	1.0350E-07	4.9929E-07	5.5901E-07
C2	5.1000E+01	8.4170E-11	4.2168E-07	5.4255E-07	2.3826E-07
I2	5.1000E+01	1.0112E-10	4.9372E-06	8.5505E-06	6.0074E-06
F2	5.1000E+01	1.6834E-11	8.4336E-08	1.0851E-07	4.7653E-08
C1	5.2000E+01	1.5628E-24	4.6070E-07	2.3039E-06	2.6557E-06
I1	5.2000E+01	3.7714E-25	1.3515E-06	1.1427E-05	2.1068E-05
F1	5.2000E+01	3.1257E-25	9.2139E-08	4.6078E-07	5.3114E-07
C2	5.2000E+01	7.1147E-11	4.1407E-07	5.4131E-07	2.4165E-07
I2	5.2000E+01	8.5470E-11	4.8481E-06	8.5310E-06	6.0927E-06
F2	5.2000E+01	1.4229E-11	8.2814E-08	1.0826E-07	4.8330E-08
C1	5.3000E+01	6.5024E-25	4.1011E-07	2.1256E-06	2.5217E-06
I1	5.3000E+01	1.5692E-25	1.2031E-06	1.0543E-05	2.0005E-05
F1	5.3000E+01	1.3005E-25	8.2022E-08	4.2513E-07	5.0433E-07
C2	5.3000E+01	6.0139E-11	4.0658E-07	5.3991E-07	2.4492E-07
I2	5.3000E+01	7.2246E-11	4.7604E-06	8.5089E-06	6.1753E-06
F2	5.3000E+01	1.2028E-11	8.1315E-08	1.0798E-07	4.8985E-08
C1	5.4000E+01	2.7054E-25	3.6508E-07	1.9606E-06	2.3929E-06
I1	5.4000E+01	6.5287E-26	1.0710E-06	9.7244E-06	1.8984E-05
F1	5.4000E+01	5.4109E-26	7.3016E-08	3.9212E-07	4.7859E-07
C2	5.4000E+01	5.0834E-11	3.9920E-07	5.3835E-07	2.4809E-07
I2	5.4000E+01	6.1068E-11	4.6741E-06	8.4843E-06	6.2551E-06

F2	5.4000E+01	1.0167E-11	7.9841E-08	1.0767E-07	4.9618E-08
C1	5.5000E+01	1.1256E-25	3.2499E-07	1.8079E-06	2.2695E-06
I1	5.5000E+01	2.7163E-26	9.5339E-07	8.9671E-06	1.8004E-05
F1	5.5000E+01	2.2513E-26	6.4998E-08	3.6158E-07	4.5389E-07
C2	5.5000E+01	4.2969E-11	3.9195E-07	5.3663E-07	2.5115E-07
I2	5.5000E+01	5.1620E-11	4.5892E-06	8.4573E-06	6.3323E-06
F2	5.5000E+01	8.5939E-12	7.8391E-08	1.0733E-07	5.0230E-08
C1	5.6000E+01	4.6833E-26	2.8931E-07	1.6667E-06	2.1512E-06
I1	5.6000E+01	1.1302E-26	8.4870E-07	8.2666E-06	1.7066E-05
F1	5.6000E+01	9.3667E-27	5.7862E-08	3.3334E-07	4.3023E-07
C2	5.6000E+01	3.6321E-11	3.8482E-07	5.3478E-07	2.5410E-07
I2	5.6000E+01	4.3633E-11	4.5057E-06	8.4281E-06	6.4068E-06
F2	5.6000E+01	7.2642E-12	7.6964E-08	1.0696E-07	5.0821E-08
C1	5.7000E+01	1.9486E-26	2.5754E-07	1.5361E-06	2.0379E-06
I1	5.7000E+01	4.7022E-27	7.5551E-07	7.6190E-06	1.6167E-05
F1	5.7000E+01	3.8971E-27	5.1508E-08	3.0722E-07	4.0758E-07
C2	5.7000E+01	3.0701E-11	3.7780E-07	5.3280E-07	2.5696E-07
I2	5.7000E+01	3.6882E-11	4.4236E-06	8.3968E-06	6.4788E-06
F2	5.7000E+01	6.1403E-12	7.5561E-08	1.0656E-07	5.1392E-08
C1	5.8000E+01	8.1073E-27	2.2926E-07	1.4155E-06	1.9296E-06
I1	5.8000E+01	1.9564E-27	6.7256E-07	7.0205E-06	1.5308E-05
F1	5.8000E+01	1.6215E-27	4.5852E-08	2.8309E-07	3.8593E-07
C2	5.8000E+01	2.5951E-11	3.7091E-07	5.3069E-07	2.5971E-07
I2	5.8000E+01	3.1176E-11	4.3428E-06	8.3636E-06	6.5482E-06
F2	5.8000E+01	5.1903E-12	7.4182E-08	1.0614E-07	5.1943E-08
C1	5.9000E+01	3.3731E-27	2.0409E-07	1.3040E-06	1.8262E-06
I1	5.9000E+01	8.1400E-28	5.9871E-07	6.4675E-06	1.4487E-05
F1	5.9000E+01	6.7463E-28	4.0818E-08	2.6079E-07	3.6524E-07
C2	5.9000E+01	2.1936E-11	3.6413E-07	5.2847E-07	2.6237E-07
I2	5.9000E+01	2.6352E-11	4.2634E-06	8.3287E-06	6.6153E-06
F2	5.9000E+01	4.3872E-12	7.2826E-08	1.0569E-07	5.2475E-08
C1	6.0000E+01	1.4034E-27	1.8168E-07	1.2010E-06	1.7274E-06
I1	6.0000E+01	3.3868E-28	5.3297E-07	5.9568E-06	1.3704E-05
F1	6.0000E+01	2.8069E-28	3.6336E-08	2.4020E-07	3.4549E-07
C2	6.0000E+01	1.8542E-11	3.5746E-07	5.2614E-07	2.6494E-07
I2	6.0000E+01	2.2275E-11	4.1854E-06	8.2920E-06	6.6800E-06
F2	6.0000E+01	3.7084E-12	7.1493E-08	1.0523E-07	5.2988E-08
C1	6.1000E+01	5.8392E-28	1.6173E-07	1.1059E-06	1.6333E-06
I1	6.1000E+01	1.4091E-28	4.7445E-07	5.4853E-06	1.2957E-05
F1	6.1000E+01	1.1678E-28	3.2346E-08	2.2119E-07	3.2666E-07
C2	6.1000E+01	1.5673E-11	3.5092E-07	5.2372E-07	2.6741E-07
I2	6.1000E+01	1.8829E-11	4.1087E-06	8.2537E-06	6.7423E-06
F2	6.1000E+01	3.1347E-12	7.0183E-08	1.0474E-07	5.3483E-08
C1	6.2000E+01	2.4295E-28	1.4397E-07	1.0182E-06	1.5436E-06
I1	6.2000E+01	5.8628E-29	4.2235E-07	5.0502E-06	1.2245E-05
F1	6.2000E+01	4.8589E-29	2.8794E-08	2.0364E-07	3.0871E-07
C2	6.2000E+01	1.3248E-11	3.4448E-07	5.2120E-07	2.6980E-07
I2	6.2000E+01	1.5915E-11	4.0334E-06	8.2140E-06	6.8024E-06
F2	6.2000E+01	2.6497E-12	6.8896E-08	1.0424E-07	5.3959E-08
C1	6.3000E+01	1.0108E-28	1.2816E-07	9.3724E-07	1.4582E-06
I1	6.3000E+01	2.4393E-29	3.7598E-07	4.6486E-06	1.1568E-05
F1	6.3000E+01	2.0216E-29	2.5633E-08	1.8745E-07	2.9163E-07
C2	6.3000E+01	1.1199E-11	3.3816E-07	5.1859E-07	2.7209E-07
I2	6.3000E+01	1.3453E-11	3.9593E-06	8.1729E-06	6.8604E-06
F2	6.3000E+01	2.2397E-12	6.7631E-08	1.0372E-07	5.4419E-08
C1	6.4000E+01	4.2056E-29	1.1409E-07	8.6256E-07	1.3769E-06
I1	6.4000E+01	1.0149E-29	3.3469E-07	4.2782E-06	1.0923E-05
F1	6.4000E+01	8.4113E-30	2.2818E-08	1.7251E-07	2.7538E-07
C2	6.4000E+01	9.4659E-12	3.3195E-07	5.1590E-07	2.7431E-07
I2	6.4000E+01	1.1372E-11	3.8866E-06	8.1305E-06	6.9162E-06
F2	6.4000E+01	1.8932E-12	6.6389E-08	1.0318E-07	5.4862E-08
C1	6.5000E+01	1.7498E-29	1.0156E-07	7.9369E-07	1.2996E-06
I1	6.5000E+01	4.2226E-30	2.9794E-07	3.9366E-06	1.0310E-05
F1	6.5000E+01	3.4996E-30	2.0313E-08	1.5874E-07	2.5992E-07
C2	6.5000E+01	8.0013E-12	3.2584E-07	5.1314E-07	2.7644E-07
I2	6.5000E+01	9.6121E-12	3.8152E-06	8.0870E-06	6.9699E-06
F2	6.5000E+01	1.6003E-12	6.5169E-08	1.0263E-07	5.5288E-08
C1	6.6000E+01	7.2803E-30	9.0412E-08	7.3019E-07	1.2262E-06
I1	6.6000E+01	1.7569E-30	2.6523E-07	3.6217E-06	9.7277E-06

F1	6.6000E+01	1.4561E-30	1.8082E-08	1.4604E-07	2.4524E-07
C2	6.6000E+01	6.7633E-12	3.1985E-07	5.1030E-07	2.7849E-07
I2	6.6000E+01	8.1249E-12	3.7450E-06	8.0423E-06	7.0216E-06
F2	6.6000E+01	1.3527E-12	6.3970E-08	1.0206E-07	5.5698E-08
C1	6.7000E+01	3.0291E-30	8.0484E-08	6.7167E-07	1.1565E-06
I1	6.7000E+01	7.3097E-31	2.3611E-07	3.3314E-06	9.1748E-06
F1	6.7000E+01	6.0581E-31	1.6097E-08	1.3433E-07	2.3130E-07
C2	6.7000E+01	5.7169E-12	3.1396E-07	5.0741E-07	2.8046E-07
I2	6.7000E+01	6.8678E-12	3.6760E-06	7.9967E-06	7.0713E-06
F2	6.7000E+01	1.1434E-12	6.2792E-08	1.0148E-07	5.6092E-08
C1	6.8000E+01	1.2603E-30	7.1647E-08	6.1773E-07	1.0904E-06
I1	6.8000E+01	3.0413E-31	2.1018E-07	3.0639E-06	8.6501E-06
F1	6.8000E+01	2.5206E-31	1.4329E-08	1.2355E-07	2.1808E-07
C2	6.8000E+01	4.8324E-12	3.0818E-07	5.0445E-07	2.8236E-07
I2	6.8000E+01	5.8052E-12	3.6083E-06	7.9500E-06	7.1191E-06
F2	6.8000E+01	9.6648E-13	6.1636E-08	1.0089E-07	5.6472E-08
C1	6.9000E+01	5.2436E-31	6.3780E-08	5.6804E-07	1.0277E-06
I1	6.9000E+01	1.2654E-31	1.8710E-07	2.8174E-06	8.1526E-06
F1	6.9000E+01	1.0487E-31	1.2756E-08	1.1361E-07	2.0553E-07
C2	6.9000E+01	4.0847E-12	3.0250E-07	5.0143E-07	2.8418E-07
I2	6.9000E+01	4.9070E-12	3.5419E-06	7.9025E-06	7.1651E-06
F2	6.9000E+01	8.1694E-13	6.0500E-08	1.0029E-07	5.6836E-08
C1	7.0000E+01	2.1817E-31	5.6777E-08	5.2228E-07	9.6822E-07
I1	7.0000E+01	5.2648E-32	1.6656E-07	2.5904E-06	7.6811E-06
F1	7.0000E+01	4.3633E-32	1.1355E-08	1.0446E-07	1.9364E-07
C2	7.0000E+01	3.4527E-12	2.9693E-07	4.9836E-07	2.8593E-07
I2	7.0000E+01	4.1478E-12	3.4766E-06	7.8541E-06	7.2092E-06
F2	7.0000E+01	6.9054E-13	5.9385E-08	9.9673E-08	5.7186E-08
C1	7.1000E+01	9.0771E-32	5.0543E-08	4.8013E-07	9.1192E-07
I1	7.1000E+01	2.1905E-32	1.4827E-07	2.3814E-06	7.2344E-06
F1	7.1000E+01	1.8154E-32	1.0109E-08	9.6025E-08	1.8238E-07
C2	7.1000E+01	2.9185E-12	2.9145E-07	4.9525E-07	2.8761E-07
I2	7.1000E+01	3.5061E-12	3.4125E-06	7.8050E-06	7.2516E-06
F2	7.1000E+01	5.8370E-13	5.8290E-08	9.9050E-08	5.7522E-08
C1	7.2000E+01	3.7767E-32	4.4993E-08	4.4132E-07	8.5861E-07
I1	7.2000E+01	9.1138E-33	1.3199E-07	2.1889E-06	6.8115E-06
F1	7.2000E+01	7.5533E-33	8.9986E-09	8.8264E-08	1.7172E-07
C2	7.2000E+01	2.4670E-12	2.8607E-07	4.9209E-07	2.8922E-07
I2	7.2000E+01	2.9636E-12	3.3495E-06	7.7552E-06	7.2922E-06
F2	7.2000E+01	4.9339E-13	5.7215E-08	9.8417E-08	5.7845E-08
C1	7.3000E+01	1.5713E-32	4.0053E-08	4.0560E-07	8.0817E-07
I1	7.3000E+01	3.7919E-33	1.1750E-07	2.0117E-06	6.4114E-06
F1	7.3000E+01	3.1427E-33	8.0105E-09	8.1119E-08	1.6163E-07
C2	7.3000E+01	2.0853E-12	2.8080E-07	4.8889E-07	2.9077E-07
I2	7.3000E+01	2.5051E-12	3.2877E-06	7.7048E-06	7.3312E-06
F2	7.3000E+01	4.1705E-13	5.6159E-08	9.7777E-08	5.8154E-08
C1	7.4000E+01	6.5377E-33	3.5655E-08	3.7272E-07	7.6047E-07
I1	7.4000E+01	1.5777E-33	1.0460E-07	1.8486E-06	6.0329E-06
F1	7.4000E+01	1.3075E-33	7.1309E-09	7.4543E-08	1.5209E-07
C2	7.4000E+01	1.7626E-12	2.7561E-07	4.8565E-07	2.9225E-07
I2	7.4000E+01	2.1175E-12	3.2270E-06	7.6538E-06	7.3685E-06
F2	7.4000E+01	3.5253E-13	5.5122E-08	9.7130E-08	5.8450E-08
C1	7.5000E+01	2.7201E-33	3.1740E-08	3.4246E-07	7.1537E-07
I1	7.5000E+01	6.5641E-34	9.3111E-08	1.6986E-06	5.6752E-06
F1	7.5000E+01	5.4402E-34	6.3480E-09	6.8492E-08	1.4307E-07
C2	7.5000E+01	1.4899E-12	2.7052E-07	4.8237E-07	2.9367E-07
I2	7.5000E+01	1.7899E-12	3.1674E-06	7.6022E-06	7.4043E-06
F2	7.5000E+01	2.9798E-13	5.4105E-08	9.6475E-08	5.8733E-08
C1	7.6000E+01	1.1317E-33	2.8255E-08	3.1462E-07	6.7276E-07
I1	7.6000E+01	2.7311E-34	8.2887E-08	1.5605E-06	5.3371E-06
F1	7.6000E+01	2.2635E-34	5.6509E-09	6.2925E-08	1.3455E-07
C2	7.6000E+01	1.2594E-12	2.6553E-07	4.7907E-07	2.9502E-07
I2	7.6000E+01	1.5129E-12	3.1089E-06	7.5501E-06	7.4385E-06
F2	7.6000E+01	2.5188E-13	5.3105E-08	9.5814E-08	5.9005E-08
C1	7.7000E+01	4.7087E-34	2.5152E-08	2.8902E-07	6.3251E-07
I1	7.7000E+01	1.1363E-34	7.3786E-08	1.4335E-06	5.0178E-06
F1	7.7000E+01	9.4175E-35	5.0304E-09	5.7803E-08	1.2650E-07
C2	7.7000E+01	1.0645E-12	2.6062E-07	4.7574E-07	2.9632E-07
I2	7.7000E+01	1.2789E-12	3.0515E-06	7.4975E-06	7.4712E-06
F2	7.7000E+01	2.1291E-13	5.2124E-08	9.5147E-08	5.9264E-08

C1	7.8000E+01	1.9591E-34	2.2390E-08	2.6546E-07	5.9451E-07
I1	7.8000E+01	4.7278E-35	6.5684E-08	1.3167E-06	4.7163E-06
F1	7.8000E+01	3.9183E-35	4.4781E-09	5.3093E-08	1.1890E-07
C2	7.8000E+01	8.9983E-13	2.5581E-07	4.7238E-07	2.9756E-07
I2	7.8000E+01	1.0810E-12	2.9951E-06	7.4446E-06	7.5024E-06
F2	7.8000E+01	1.7997E-13	5.1161E-08	9.4475E-08	5.9512E-08
C1	7.9000E+01	8.1513E-35	1.9932E-08	2.4380E-07	5.5865E-07
I1	7.9000E+01	1.9670E-35	5.8472E-08	1.2092E-06	4.4318E-06
F1	7.9000E+01	1.6303E-35	3.9864E-09	4.8761E-08	1.1173E-07
C2	7.9000E+01	7.6061E-13	2.5108E-07	4.6899E-07	2.9874E-07
I2	7.9000E+01	9.1374E-13	2.9398E-06	7.3913E-06	7.5322E-06
F2	7.9000E+01	1.5212E-13	5.0216E-08	9.3798E-08	5.9748E-08
C1	8.0000E+01	3.3914E-35	1.7743E-08	2.2389E-07	5.2482E-07
I1	8.0000E+01	8.1842E-36	5.2051E-08	1.1105E-06	4.1635E-06
F1	8.0000E+01	6.7829E-36	3.5487E-09	4.4778E-08	1.0496E-07
C2	8.0000E+01	6.4293E-13	2.4644E-07	4.6559E-07	2.9987E-07
I2	8.0000E+01	7.7236E-13	2.8854E-06	7.3376E-06	7.5606E-06
F2	8.0000E+01	1.2859E-13	4.9288E-08	9.3117E-08	5.9973E-08
C1	8.1000E+01	1.4111E-35	1.5795E-08	2.0558E-07	4.9292E-07
I1	8.1000E+01	3.4051E-36	4.6336E-08	1.0197E-06	3.9104E-06
F1	8.1000E+01	2.8221E-36	3.1590E-09	4.1116E-08	9.8583E-08
C2	8.1000E+01	5.4345E-13	2.4188E-07	4.6216E-07	3.0094E-07
I2	8.1000E+01	6.5286E-13	2.8321E-06	7.2836E-06	7.5876E-06
F2	8.1000E+01	1.0869E-13	4.8377E-08	9.2433E-08	6.0187E-08
C1	8.2000E+01	5.8709E-36	1.4061E-08	1.8875E-07	4.6284E-07
I1	8.2000E+01	1.4167E-36	4.1248E-08	9.3619E-07	3.6718E-06
F1	8.2000E+01	1.1742E-36	2.8122E-09	3.7750E-08	9.2569E-08
C2	8.2000E+01	4.5937E-13	2.3741E-07	4.5872E-07	3.0196E-07
I2	8.2000E+01	5.5185E-13	2.7797E-06	7.2294E-06	7.6133E-06
F2	8.2000E+01	9.1874E-14	4.7482E-08	9.1744E-08	6.0391E-08
C1	8.3000E+01	2.4427E-36	1.2517E-08	1.7328E-07	4.3451E-07
I1	8.3000E+01	5.8946E-37	3.6719E-08	8.5947E-07	3.4470E-06
F1	8.3000E+01	4.8853E-37	2.5034E-09	3.4657E-08	8.6901E-08
C2	8.3000E+01	3.8830E-13	2.3302E-07	4.5527E-07	3.0292E-07
I2	8.3000E+01	4.6647E-13	2.7284E-06	7.1749E-06	7.6376E-06
F2	8.3000E+01	7.7659E-14	4.6604E-08	9.1053E-08	6.0585E-08
C1	8.4000E+01	1.0163E-36	1.1142E-08	1.5907E-07	4.0781E-07
I1	8.4000E+01	2.4525E-37	3.2687E-08	7.8897E-07	3.2353E-06
F1	8.4000E+01	2.0326E-37	2.2285E-09	3.1814E-08	8.1563E-08
C2	8.4000E+01	3.2822E-13	2.2871E-07	4.5180E-07	3.0384E-07
I2	8.4000E+01	3.9430E-13	2.6779E-06	7.1202E-06	7.6608E-06
F2	8.4000E+01	6.5644E-14	4.5743E-08	9.0359E-08	6.0768E-08
C1	8.5000E+01	4.2285E-37	9.9190E-09	1.4601E-07	3.8268E-07
I1	8.5000E+01	1.0204E-37	2.9098E-08	7.2419E-07	3.0358E-06
F1	8.5000E+01	8.4569E-38	1.9838E-09	2.9202E-08	7.6535E-08
C2	8.5000E+01	2.7744E-13	2.2448E-07	4.4831E-07	3.0471E-07
I2	8.5000E+01	3.3329E-13	2.6284E-06	7.0654E-06	7.6826E-06
F2	8.5000E+01	5.5487E-14	4.4897E-08	8.9663E-08	6.0941E-08
C1	8.6000E+01	1.7593E-37	8.8299E-09	1.3401E-07	3.5901E-07
I1	8.6000E+01	4.2455E-38	2.5903E-08	6.6467E-07	2.8481E-06
F1	8.6000E+01	3.5186E-38	1.7660E-09	2.6802E-08	7.1802E-08
C2	8.6000E+01	2.3451E-13	2.2033E-07	4.4482E-07	3.0553E-07
I2	8.6000E+01	2.8172E-13	2.5798E-06	7.0104E-06	7.7033E-06
F2	8.6000E+01	4.6902E-14	4.4066E-08	8.8965E-08	6.1105E-08
C1	8.7000E+01	7.3198E-38	7.8604E-09	1.2299E-07	3.3674E-07
I1	8.7000E+01	1.7664E-38	2.3059E-08	6.0999E-07	2.6714E-06
F1	8.7000E+01	1.4640E-38	1.5721E-09	2.4597E-08	6.7349E-08
C2	8.7000E+01	1.9823E-13	2.1626E-07	4.4132E-07	3.0630E-07
I2	8.7000E+01	2.3813E-13	2.5320E-06	6.9552E-06	7.7228E-06
F2	8.7000E+01	3.9646E-14	4.3251E-08	8.8265E-08	6.1260E-08
C1	8.8000E+01	3.0455E-38	6.9973E-09	1.1286E-07	3.1579E-07
I1	8.8000E+01	7.3494E-39	2.0527E-08	5.5977E-07	2.5052E-06
F1	8.8000E+01	6.0910E-39	1.3995E-09	2.2572E-08	6.3158E-08
C2	8.8000E+01	1.6756E-13	2.1225E-07	4.3782E-07	3.0703E-07
I2	8.8000E+01	2.0129E-13	2.4852E-06	6.9000E-06	7.7411E-06
F2	8.8000E+01	3.3512E-14	4.2451E-08	8.7564E-08	6.1405E-08
C1	8.9000E+01	1.2671E-38	6.2290E-09	1.0356E-07	2.9609E-07
I1	8.9000E+01	3.0578E-39	1.8273E-08	5.1364E-07	2.3489E-06
F1	8.9000E+01	2.5343E-39	1.2458E-09	2.0712E-08	5.9217E-08

C2	8.9000E+01	1.4163E-13	2.0833E-07	4.3431E-07	3.0771E-07
I2	8.9000E+01	1.7015E-13	2.4392E-06	6.8447E-06	7.7583E-06
F2	8.9000E+01	2.8327E-14	4.1666E-08	8.6862E-08	6.1542E-08
C1	9.0000E+01	5.2720E-39	5.5450E-09	9.5019E-08	2.7756E-07
I1	9.0000E+01	1.2722E-39	1.6267E-08	4.7128E-07	2.2019E-06
F1	9.0000E+01	1.0544E-39	1.1090E-09	1.9004E-08	5.5511E-08
C2	9.0000E+01	1.1972E-13	2.0447E-07	4.3080E-07	3.0835E-07
I2	9.0000E+01	1.4382E-13	2.3941E-06	6.7893E-06	7.7744E-06
F2	9.0000E+01	2.3944E-14	4.0895E-08	8.6159E-08	6.1669E-08
C1	9.1000E+01	2.1935E-39	4.9361E-09	8.7177E-08	2.6014E-07
I1	9.1000E+01	5.2933E-40	1.4481E-08	4.3239E-07	2.0637E-06
F1	9.1000E+01	4.3870E-40	9.8723E-10	1.7435E-08	5.2028E-08
C2	9.1000E+01	1.0120E-13	2.0069E-07	4.2728E-07	3.0894E-07
I2	9.1000E+01	1.2157E-13	2.3498E-06	6.7339E-06	7.7894E-06
F2	9.1000E+01	2.0239E-14	4.0138E-08	8.5456E-08	6.1788E-08
C1	9.2000E+01	9.1264E-40	4.3941E-09	7.9976E-08	2.4377E-07
I1	9.2000E+01	2.2024E-40	1.2891E-08	3.9667E-07	1.9339E-06
F1	9.2000E+01	1.8253E-40	8.7883E-10	1.5995E-08	4.8754E-08
C2	9.2000E+01	8.5539E-14	1.9698E-07	4.2376E-07	3.0950E-07
I2	9.2000E+01	1.0276E-13	2.3063E-06	6.6784E-06	7.8034E-06
F2	9.2000E+01	1.7108E-14	3.9395E-08	8.4752E-08	6.1899E-08
C1	9.3000E+01	3.7972E-40	3.9117E-09	7.3365E-08	2.2839E-07
I1	9.3000E+01	9.1632E-41	1.1475E-08	3.6388E-07	1.8119E-06
F1	9.3000E+01	7.5943E-41	7.8233E-10	1.4673E-08	4.5679E-08
C2	9.3000E+01	7.2305E-14	1.9333E-07	4.2024E-07	3.1001E-07
I2	9.3000E+01	8.6861E-14	2.2636E-06	6.6230E-06	7.8163E-06
F2	9.3000E+01	1.4461E-14	3.8666E-08	8.4049E-08	6.2002E-08
C1	9.4000E+01	1.5799E-40	3.4822E-09	6.7297E-08	2.1395E-07
I1	9.4000E+01	3.8125E-41	1.0215E-08	3.3378E-07	1.6973E-06
F1	9.4000E+01	3.1597E-41	6.9643E-10	1.3459E-08	4.2790E-08
C2	9.4000E+01	6.1117E-14	1.8975E-07	4.1673E-07	3.1048E-07
I2	9.4000E+01	7.3422E-14	2.2218E-06	6.5676E-06	7.8282E-06
F2	9.4000E+01	1.2223E-14	3.7951E-08	8.3346E-08	6.2096E-08
C1	9.5000E+01	6.5732E-41	3.0998E-09	6.1726E-08	2.0039E-07
I1	9.5000E+01	1.5862E-41	9.0935E-09	3.0615E-07	1.5897E-06
F1	9.5000E+01	1.3146E-41	6.1996E-10	1.2345E-08	4.0077E-08
C2	9.5000E+01	5.1661E-14	1.8624E-07	4.1321E-07	3.1091E-07
I2	9.5000E+01	6.2062E-14	2.1806E-06	6.5122E-06	7.8391E-06
F2	9.5000E+01	1.0332E-14	3.7249E-08	8.2643E-08	6.2183E-08
C1	9.6000E+01	2.7349E-41	2.7594E-09	5.6613E-08	1.8765E-07
I1	9.6000E+01	6.5997E-42	8.0950E-09	2.8080E-07	1.4887E-06
F1	9.6000E+01	5.4697E-42	5.5189E-10	1.1323E-08	3.7530E-08
C2	9.6000E+01	4.3668E-14	1.8280E-07	4.0970E-07	3.1131E-07
I2	9.6000E+01	5.2459E-14	2.1403E-06	6.4569E-06	7.8491E-06
F2	9.6000E+01	8.7337E-15	3.6559E-08	8.1941E-08	6.2262E-08
C1	9.7000E+01	1.1379E-41	2.4564E-09	5.1921E-08	1.7570E-07
I1	9.7000E+01	2.7459E-42	7.2061E-09	2.5752E-07	1.3938E-06
F1	9.7000E+01	2.2758E-42	4.9129E-10	1.0384E-08	3.5140E-08
C2	9.7000E+01	3.6912E-14	1.7941E-07	4.0620E-07	3.1167E-07
I2	9.7000E+01	4.4343E-14	2.1007E-06	6.4016E-06	7.8581E-06
F2	9.7000E+01	7.3824E-15	3.5883E-08	8.1240E-08	6.2333E-08
C1	9.8000E+01	4.7343E-42	2.1867E-09	4.7615E-08	1.6448E-07
I1	9.8000E+01	1.1425E-42	6.4149E-09	2.3616E-07	1.3049E-06
F1	9.8000E+01	9.4686E-43	4.3734E-10	9.5230E-09	3.2896E-08
C2	9.8000E+01	3.1201E-14	1.7609E-07	4.0270E-07	3.1199E-07
I2	9.8000E+01	3.7482E-14	2.0618E-06	6.3465E-06	7.8661E-06
F2	9.8000E+01	6.2402E-15	3.5219E-08	8.0540E-08	6.2397E-08
C1	9.9000E+01	1.9698E-42	1.9466E-09	4.3663E-08	1.5396E-07
I1	9.9000E+01	4.7534E-43	5.7105E-09	2.1657E-07	1.2214E-06
F1	9.9000E+01	3.9395E-43	3.8932E-10	8.7327E-09	3.0791E-08
C2	9.9000E+01	2.6373E-14	1.7283E-07	3.9920E-07	3.1227E-07
I2	9.9000E+01	3.1683E-14	2.0236E-06	6.2914E-06	7.8733E-06
F2	9.9000E+01	5.2747E-15	3.4567E-08	7.9841E-08	6.2454E-08
C1	1.0000E+02	8.1955E-43	1.7329E-09	4.0038E-08	1.4408E-07
I1	1.0000E+02	1.9777E-43	5.0835E-09	1.9858E-07	1.1431E-06
F1	1.0000E+02	1.6391E-43	3.4657E-10	8.0075E-09	2.8817E-08
C2	1.0000E+02	2.2293E-14	1.6963E-07	3.9572E-07	3.1252E-07
I2	1.0000E+02	2.6781E-14	1.9862E-06	6.2364E-06	7.8796E-06
F2	1.0000E+02	4.4586E-15	3.3927E-08	7.9143E-08	6.2504E-08

Verification Problem 4

Parameter Definition File

```
Verification problem #4 for FOLAT using solubility limited release
'verify4.out'
'verify2.pcp'
'verify2.inv'
1.0e-6  .0001    1.0e-30              eps h1 hmin
3   1  1                            mlayer  nprog  nmat
'verify2'                          rel file name
138                                 mw
1.                                   sol
100.                                thalf
10000.  0   0                           initial y(i)
.1 .5  1.0                               kd(i,j)
0. 0. 0.                           kx(i.j)
1 3
$ layer 1-3
1  1.5  10  10                     thick(1),rho(1) len(1) width(1)
1710,0.2724,0.0321,7.51,2.298      sk(1),ths(1),thr(1),alpha(1),rn(1)
$ output times
1                                  ntimes
0. 466. 2.0                          t1,t2,tp
```

Water Flux File- Same as Verification Problem 2

Release File- Same as Verification Problem 2

Output File

Note: Output is truncated after 60 years

```
   ****************************************************
   *                                                  *
   *     This output was produced by the model:       *
   *                                                  *
   *                      FOLAT                       *
   *    The First-Order-Leach-And-Transport model.    *
   *    A general purpose solver for leaching and     *
   *    subsurface transport of radionuclides in      *
   *  surface or buried locations. Version date:      *
   *                     111602                       *
   *                  Arthur S. Rood                  *
   *                    K-Spar Inc                    *
   *              493 N 4154 E Rigby ID 83442         *
   *                    asr@srv.net                   *
   ****************************************************
 Date: 11/16/2002 Time: 20:19:48.600
 Input File:          verify4.par
 Output File:         verify4.out
 Pecipitation File:   verify2.pcp
 Release File:        verify2.inv
 Number of layers:           3
 Number of progeny           1
 Nuclide Names             verify
 Half lifes (y)             1.000E+02
 Solubility (mg/L)          1.000E+00
 Molar Weight (g/mol)       1.380E+02
 Kd Value for Member 1 for Each Layer (mL/g)
  1.000E-01  5.000E-01  1.000E+00
 Kx Value for Member 1 for Each Layer (1/y)
  0.000E+00  0.000E+00  0.000E+00
 Initial Activity for Member 1 for Each Layer (Ci)
```

```
 1.000E+04  0.000E+00  0.000E+00
Thickness of each layer (m)
 1.000E+00  1.000E+00  1.000E+00
Bulk Density (g/cm**3)
 1.500E+00  1.500E+00  1.500E+00
K-sat (m/y)
 1.710E+03  1.710E+03  1.710E+03
Porosity
 2.724E-01  2.724E-01  2.724E-01
Residual Moisture Content
 3.210E-02  3.210E-02  3.210E-02
Alpha (1/m)
 7.510E+00  7.510E+00  7.510E+00
Van Genuchten n
 2.298E+00  2.298E+00  2.298E+00
Length (m)
 1.000E+01  1.000E+01  1.000E+01
Width (m)
 1.000E+01  1.000E+01  1.000E+01
----------------------
  Calculated Values
----------------------
Moisture Content in each Layer at each Time
 0.000E+00  6.060E-02  5.613E-02  5.235E-02
 1.000E+05  6.060E-02  5.613E-02  5.235E-02
Decay Constants of each Member (1/y)
 6.931E-03
Solubility of Each Member (Ci/m**3)
 2.593E+01
Initial Pore Water Conconcentration for Member 1 for Each Layer (Ci/m**3)
 4.748E+02  0.000E+00  0.000E+00
Initial Leach Rate for Member 1 for Each Layer (1/y)
 4.748E-01  6.203E-02  1.610E-02
Conversion factors from activity (Ci) to mass (atoms)
 1.683E+20
Conversion factors from mass (atoms) to activity (Ci)
 5.940E-21
----------------------
 Start of Calculation
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C=Concentration (Ci/m**3), I=Inventory (Ci) F=Flux (Ci/y)

ID  Time (y)       Layer 1     Layer 2     Layer 3
C1  0.0000E+00  4.7483E+02  0.0000E+00  0.0000E+00
I1  0.0000E+00  1.0000E+04  0.0000E+00  0.0000E+00
F1  0.0000E+00  4.7483E+03  0.0000E+00  0.0000E+00
C1  2.0000E+00  2.5927E+01  6.0086E+00  1.9494E-01
I1  2.0000E+00  9.3474E+03  4.8437E+02  3.0262E+01
F1  2.0000E+00  2.5927E+02  3.0043E+01  4.8735E-01
C1  4.0000E+00  2.5927E+01  1.1243E+01  7.3427E-01
I1  4.0000E+00  8.7037E+03  9.0634E+02  1.1399E+02
F1  4.0000E+00  2.5927E+02  5.6216E+01  1.8357E+00
C1  6.0000E+00  2.5927E+01  1.5803E+01  1.5570E+00
I1  6.0000E+00  8.0689E+03  1.2740E+03  2.4170E+02
F1  6.0000E+00  2.5927E+02  7.9017E+01  3.8925E+00
C1  8.0000E+00  2.5927E+01  1.9776E+01  2.6107E+00
I1  8.0000E+00  7.4429E+03  1.5942E+03  4.0527E+02
F1  8.0000E+00  2.5927E+02  9.8881E+01  6.5268E+00
C1  1.0000E+01  2.5927E+01  2.3237E+01  3.8505E+00
I1  1.0000E+01  6.8255E+03  1.8732E+03  5.9773E+02
F1  1.0000E+01  2.5927E+02  1.1619E+02  9.6262E+00
C1  1.2000E+01  2.5927E+01  2.5927E+01  5.2367E+00
I1  1.2000E+01  6.2165E+03  2.1164E+03  8.1291E+02
F1  1.2000E+01  2.5927E+02  1.2963E+02  1.3092E+01
C1  1.4000E+01  2.5927E+01  2.5927E+01  6.6332E+00
I1  1.4000E+01  5.6160E+03  2.3448E+03  1.0297E+03
F1  1.4000E+01  2.5927E+02  1.2963E+02  1.6583E+01
C1  1.6000E+01  2.5927E+01  2.5927E+01  7.9668E+00
I1  1.6000E+01  5.0237E+03  2.5700E+03  1.2367E+03
F1  1.6000E+01  2.5927E+02  1.2963E+02  1.9917E+01
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C1	1.8000E+01	2.5927E+01	2.5927E+01	9.2403E+00
I1	1.8000E+01	4.4396E+03	2.7921E+03	1.4344E+03
F1	1.8000E+01	2.5927E+02	1.2963E+02	2.3101E+01
C1	2.0000E+01	2.5927E+01	2.5927E+01	1.0457E+01
I1	2.0000E+01	3.8635E+03	3.0111E+03	1.6232E+03
F1	2.0000E+01	2.5927E+02	1.2963E+02	2.6141E+01
C1	2.2000E+01	2.5927E+01	2.5927E+01	1.1618E+01
I1	2.2000E+01	3.2953E+03	3.2271E+03	1.8035E+03
F1	2.2000E+01	2.5927E+02	1.2963E+02	2.9045E+01
C1	2.4000E+01	2.5927E+01	2.5927E+01	1.2727E+01
I1	2.4000E+01	2.7350E+03	3.4402E+03	1.9757E+03
F1	2.4000E+01	2.5927E+02	1.2963E+02	3.1818E+01
C1	2.6000E+01	2.5927E+01	2.5927E+01	1.3786E+01
I1	2.6000E+01	2.1824E+03	3.6503E+03	2.1401E+03
F1	2.6000E+01	2.5927E+02	1.2963E+02	3.4466E+01
C1	2.8000E+01	2.5927E+01	2.5927E+01	1.4798E+01
I1	2.8000E+01	1.6374E+03	3.8575E+03	2.2972E+03
F1	2.8000E+01	2.5927E+02	1.2963E+02	3.6995E+01
C1	3.0000E+01	2.5927E+01	2.5927E+01	1.5764E+01
I1	3.0000E+01	1.0999E+03	4.0619E+03	2.4471E+03
F1	3.0000E+01	2.5927E+02	1.2963E+02	3.9410E+01
C1	3.2000E+01	2.5927E+01	2.5927E+01	1.6686E+01
I1	3.2000E+01	5.6979E+02	4.2635E+03	2.5903E+03
F1	3.2000E+01	2.5927E+02	1.2963E+02	4.1716E+01
C1	3.4000E+01	1.0332E+01	2.5927E+01	1.7567E+01
I1	3.4000E+01	2.1760E+02	4.2916E+03	2.7271E+03
F1	3.4000E+01	1.0332E+02	1.2963E+02	4.3918E+01
C1	3.6000E+01	3.9423E+00	2.5927E+01	1.8409E+01
I1	3.6000E+01	8.3025E+01	4.1067E+03	2.8577E+03
F1	3.6000E+01	3.9423E+01	1.2963E+02	4.6022E+01
C1	3.8000E+01	1.5042E+00	2.5927E+01	1.9212E+01
I1	3.8000E+01	3.1678E+01	3.8428E+03	2.9824E+03
F1	3.8000E+01	1.5042E+01	1.2963E+02	4.8030E+01
C1	4.0000E+01	5.7391E-01	2.5927E+01	1.9979E+01
I1	4.0000E+01	1.2087E+01	3.5516E+03	3.1015E+03
F1	4.0000E+01	5.7391E+00	1.2963E+02	4.9948E+01
C1	4.2000E+01	2.1898E-01	2.5927E+01	2.0712E+01
I1	4.2000E+01	4.6117E+00	3.2525E+03	3.2152E+03
F1	4.2000E+01	2.1898E+00	1.2963E+02	5.1780E+01
C1	4.4000E+01	8.3550E-02	2.5927E+01	2.1412E+01
I1	4.4000E+01	1.7596E+00	2.9531E+03	3.3238E+03
F1	4.4000E+01	8.3550E-01	1.2963E+02	5.3529E+01
C1	4.6000E+01	3.1879E-02	2.5927E+01	2.2080E+01
I1	4.6000E+01	6.7138E-01	2.6560E+03	3.4276E+03
F1	4.6000E+01	3.1879E-01	1.2963E+02	5.5199E+01
C1	4.8000E+01	1.2163E-02	2.5927E+01	2.2718E+01
I1	4.8000E+01	2.5616E-01	2.3624E+03	3.5266E+03
F1	4.8000E+01	1.2163E-01	1.2963E+02	5.6795E+01
C1	5.0000E+01	4.6409E-03	2.5710E+01	2.3327E+01
I1	5.0000E+01	9.7739E-02	2.0726E+03	3.6211E+03
F1	5.0000E+01	4.6409E-02	1.2855E+02	5.8317E+01
C1	5.2000E+01	1.7707E-03	2.2399E+01	2.3788E+01
I1	5.2000E+01	3.7292E-02	1.8056E+03	3.6927E+03
F1	5.2000E+01	1.7707E-02	1.1199E+02	5.9469E+01
C1	5.4000E+01	6.7562E-04	1.9514E+01	2.4033E+01
I1	5.4000E+01	1.4229E-02	1.5730E+03	3.7308E+03
F1	5.4000E+01	6.7562E-03	9.7568E+01	6.0083E+01
C1	5.6000E+01	2.5778E-04	1.7000E+01	2.4098E+01
I1	5.6000E+01	5.4290E-03	1.3704E+03	3.7408E+03
F1	5.6000E+01	2.5778E-03	8.4999E+01	6.0245E+01
C1	5.8000E+01	9.8357E-05	1.4810E+01	2.4012E+01
I1	5.8000E+01	2.0714E-03	1.1939E+03	3.7275E+03
F1	5.8000E+01	9.8357E-04	7.4049E+01	6.0030E+01
C1	6.0000E+01	3.7528E-05	1.2902E+01	2.3801E+01
I1	6.0000E+01	7.9036E-04	1.0401E+03	3.6948E+03
F1	6.0000E+01	3.7528E-04	6.4510E+01	5.9504E+01