APPENDIX II

Radiological Risk Assessment

Commercial Low-Level Radioactive Waste Disposal Site

Richland, Washington

Washington State Department of Health
Office of Radiation Protection

RADIOLOGICAL RISK ASSESSMENT

LOW-LEVEL RADIOACTIVE WASTE DISPOSAL SITE
RICHLAND, WASHINGTON

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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
<u>Enhanced Designs:</u> **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

			Rural Resident Scenario	Hanford Guidance[5]	HSRAM	MTCA[6]
Media/Pathway		Exposure Parameters	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

[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.

		Rural Resident Scenario	Hanford Guidance[5]	HSRAM	MTCA[6]
Media/Pathway	Exposure Parameters	Exposure/Intake/Contact Rate			

	Dermal	Dermal exposure rate (mg) (child)	NA	NC	ND	ND
		(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.

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

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 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

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

be 10 percent of the intake rate of other animal tissue.[14] Note, 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)	1.0×10^{-4}	1.0×10^{-4}	1×10^{-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)

[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 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**			
		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
		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.

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[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

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

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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.

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%).

surface is conservatively assumed to sufficiently encompass the perimeter of the house, thereby contributing to an indoor dose from external radiation.

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 m²) 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]

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[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.

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 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

[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.

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.

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

[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.

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 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
	Inhalation	Air mass loading (ug/m3)	100

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**
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.

[34] 165 root +178 leafy).

(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.
(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:

  <u>Equation 1</u>

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

  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

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

[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.

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

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

Where:

[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.

- $Dose_{sweat\ lodge}$ = 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 $1.5x10^{-5}$ 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]:

<u>Equation 5</u>

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

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 per

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[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.

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 2.2×10^{-3} pCi/cm^2. This contaminant concentration would need to be at least nine (9) orders of magnitude greater before deterministic risks such as skin burns became an issue.[50] Dermal exposure for radionuclides is therefore not included in this analysis.

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[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.

[50] Based upon the NCRP-recommended limit of 75µCi-hrs of exposure [NCRP, 1989].

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

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[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].

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.

4.4.1.1.1 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:

<u>Equation 9</u>

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

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]:

  <u>Equation 10</u>

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

  Where:

- C_i = Indoor concentration (pCi/l)

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[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/m²*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:

  <u>Equation 11</u>

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

- 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:

  <u>Equation 12</u>

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

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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

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

Where:

- $C_{\text{Radon in air}}$ = 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

[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.

contamination. For the gaseous emitters, this is the 228,000-square meter area of the 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:

<u>Equation 14</u>

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

Where:

- C_a = Air concentration offsite (pCi/l)
- Radon Flux = ground level emission rate (pCi/m²*s)
- $Area_{site}$ = Area of trenches (m²)
- 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 6.4×10^{-6} Ci/m²y for the realistic estimate, and 10.7×10^{-6} Ci/m²y for the conservative case.[63]

[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.

- 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 (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

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

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

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

Where:

- $C_{C\text{-}14,v}$ = Concentration of carbon 14 in plants (pCi/kg)

[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.

- $C_{C,v}$ = Fraction of stable carbon in plants[66] (0.1)
- 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-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:

<u>Equation 17</u>

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

[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, 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.

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 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]:

<u>Equation 18</u>

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

[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.

Where:

- $C_{H\text{-}3\ in\ air}$ = 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)

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[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:

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]:

<u>Equation 19</u>

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

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:

<u>Equation 20</u>

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

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.

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.

<u>Equation 21</u>

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

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.

<u>Equation 22</u>

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

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

[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.

Where:

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

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)

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

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

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

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

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

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]

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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 m² 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 m² 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 m² 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-m² area is expected to also contain contamination as a result of contaminated irrigation water.

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

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].

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

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.

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:

<u>Equation 30</u>

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

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

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

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.

[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.

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)

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

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

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)

- $W_{D\text{-}W}$ = 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

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

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.

[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.

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]

- 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

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

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 sweat lodge calculations are considered a worst case estimate of the potential exposure to contaminants. The operating assumption is that 100% of the contaminants in the groundwater (used as the source of steam for the sweat lodge) will go 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 contaminate 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 go airborne, further reducing the air concentration from those used in the calculation. Until data is 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

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

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 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

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[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.
[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.

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

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

[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.

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.

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.

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

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

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)

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

Cover	Enhanced GeoSynthetic with Construction in 2056			Enhanced GeoSynthetic			Enhanced GeoSynthetic		
Closure Date	2056			2005			2215		
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	70	105	92	70	106	93	28	97	88
Rural Resident Child	68	102	89	63	103	91	25	94	86
Native American Adult	171	104	95	90	107	104	44	101	101
Native American Child	82	95	87	71	95	89	30	88	86
Offsite Resident									
Rural Resident Adult	36	2	5	9	3	6	8	2	7
Rural Resident Child	39	2	6	9	2	7	8	2	9
Native American Adult	130	7	13	19	11	21	18	11	24
Native American Child	48	3	9	10	4	11	9	3	14
Onsite Upland Hunter									
Native American Adult									
Native American Child									
Resident River									
Native American Adult									
Native American Child									

Table 5.1.1 Dose Estimate for All Covers and Scenarios (mrem/y) (continued)

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) (continued)

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

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
Scenarios									
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 5.1.2 Lifetime Cancer Risk

Scenarios	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 (continued)

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 (continued)

	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 (continued)

	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 (continued)

	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 (continued)

	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 (continued)

	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 (continued)

	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 Parametric 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. These single point estimates of the predicted dose or risk than an individual will receive in the future is an unknown quantity and therefore subject to uncertainty. In order to examine the overall variability of the dose endpoint, probability distributions are quantitatively defined that are intended to incorporate the "true" range[86] for a given variable. Once the distributions are defined for the parameters of interest[87], an overall dose and risk model is run using a Monte Carlo approach. This approach allows each parameter specified to vary within a probability 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 parameter 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.

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[86] Estimates for probability distributions for any given parameter are termed subjective distributions as they are limited by the state of knowledge about a given variable and are therefore subject to some limitations.

[87] Those parameters that have a significant effect on the predicted outcome are identified through a sensitivity analysis.

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.[88] 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.

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

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

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.[89] The resulting range is

[89] 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

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].[90]

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.[91]

- 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.

- 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

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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].

[90] 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.

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

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 $\pm$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.[92] 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 the radiological analysis there are significantly more pathways considered, the graph does provide the peak doses and overall uncertainty as time progresses.

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

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

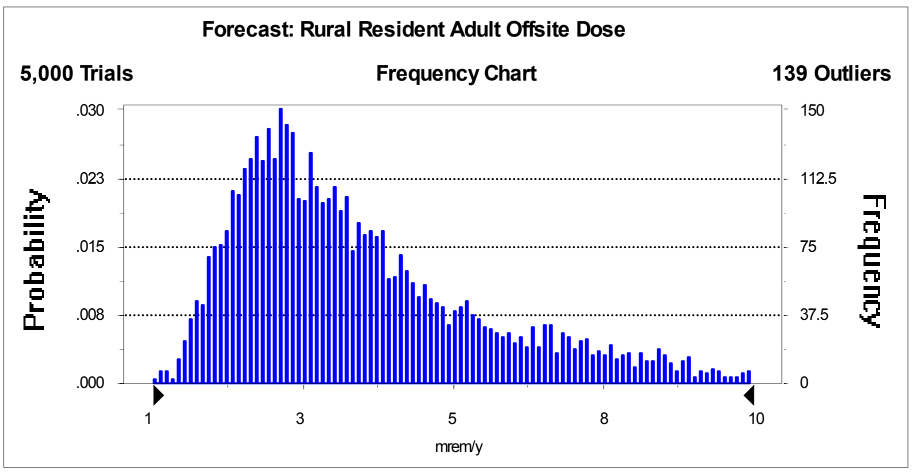
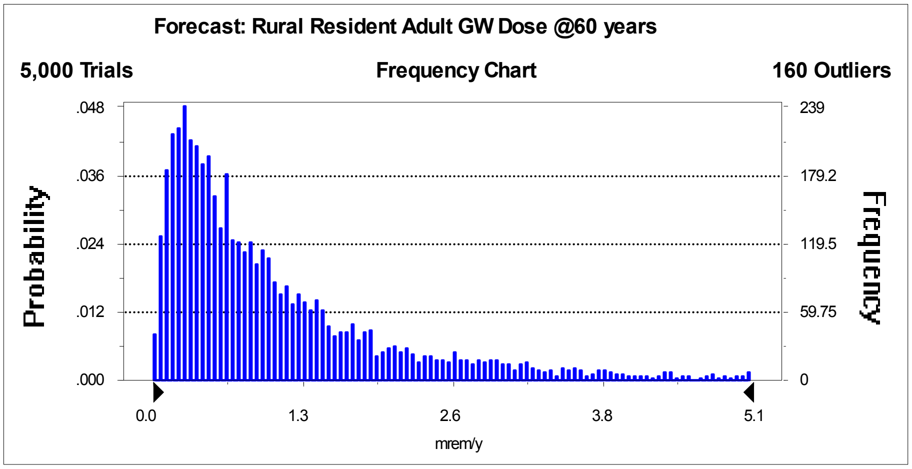


Figure 6.6.2 is a frequency distribution chart showing the groundwater contribution for all radionuclides, with the exception of tritium. The mean, median, and modal values for all groundwater dose contributors, excluding tritium, are all less than 2 mrem/y. The difference between Figures 6.6.1 and 6.6.2 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).[93] This correction factor was applied to all tritium estimates, as the error is assumed constant.[94] 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



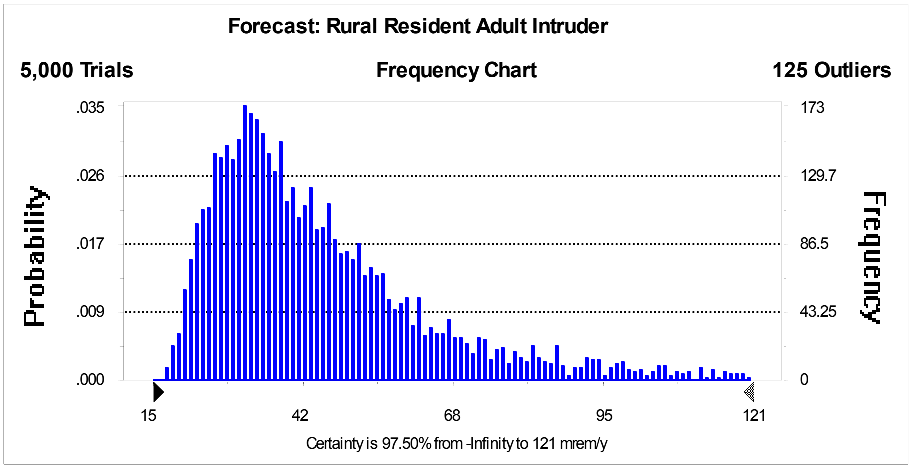
6.6.2 Estimated Dose Distributions at 1000 Years Post-Closure

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

[93] 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.
[94] 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

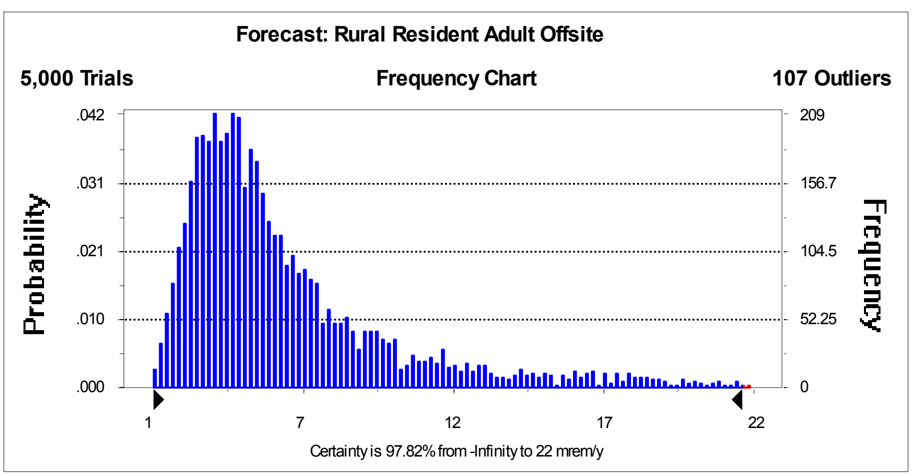


All of the figures for the uncertainty analysis are log-normally distributed and are positively skewed to the right. This distribution graphically reinforces the limited probability that the upper bound estimates represent a likely exposure event.

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

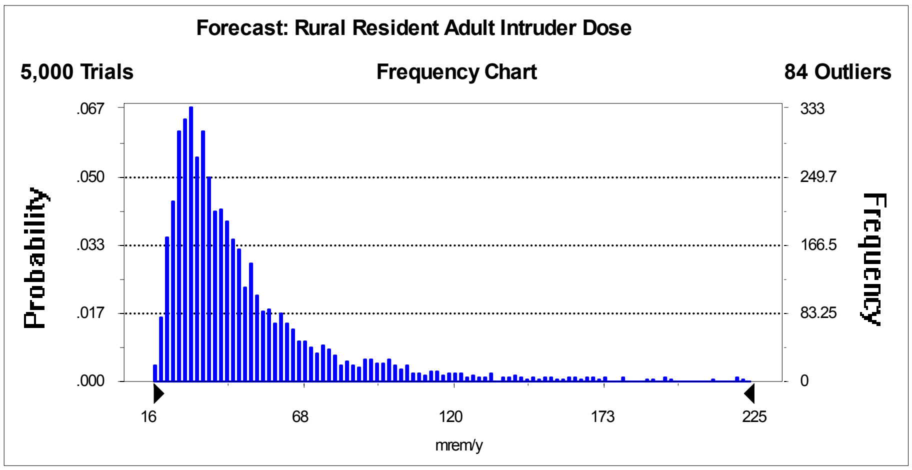


6.6.3 Estimated Dose Distributions at 10,000 Years Post-Closure

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

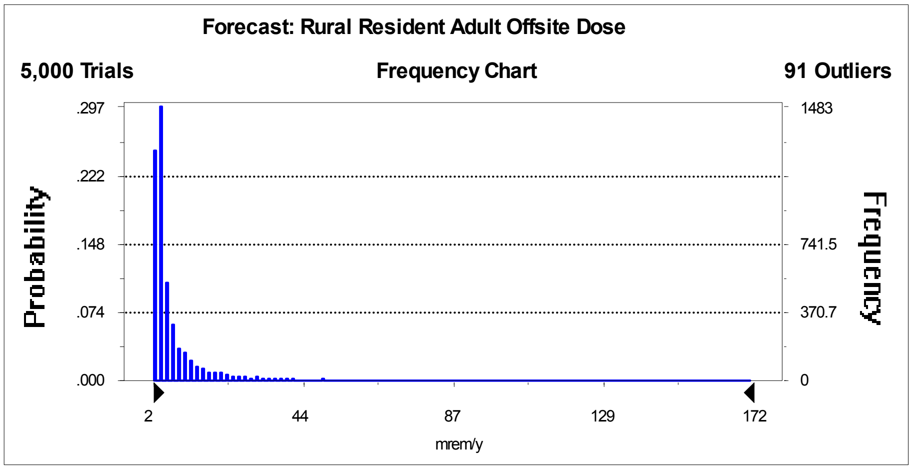


The upper bound value reflects the increased uncertainty associated with projections so far into the future and is further discussed in the Groundwater Appendix. The large variation between an upper bound estimate and the most likely value also indicates the impact of lifestyle assumptions and patterns. 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 parameter 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[95] 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.

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[95] 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 likely 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%.[96] 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 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.

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[96] 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.

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 parameter uncertainty for the rural resident adult. The parameter subjective probability distributions are provided in Attachment 1. Further uncertainties for a given parameter 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 parametric 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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