Pacific Northwest National Laboratory

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Analysis of the Long-Term Impacts of TRU Waste Remaining at Generator/Storage Sites for No Action Alternative 2

Support Information for the Waste Isolation Pilot Plant Disposal-
Phase Final Supplemental Environmental Impact Statement

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either inhalation or groundwater ingestion for all sites. Transuranic radionuclides (Pu-238,-239,-240, and Am-241) were major contributors except at INEEL, where C-14 was the major contributor.

The estimated hazardous chemical carcinogenic impacts to the MEI during the maximum 70-year lifetime over the 10,000-year analysis period ranged from a 5.4 E-03 probability of cancer incidence at INEEL down to a 1.3 E-07 probability of cancer incidence at LANL. At INEEL, the maximum probability of cancer incidence was estimated to occur after about 3,185 years (the 45th consecutive 70-year lifetime), while the maximum occurred much earlier at all other sites. The dominant exposure pathway was groundwater ingestion at all sites except LANL, where it was resuspended soil ingestion. The key contaminant was 1,1,2,2-tetrachloroethane for MEI exposure at all sites.

The estimated hazardous chemical noncarcinogenic impacts to the MEI during the maximum 70-year lifetime over the 10,000-year analysis period ranged from a hazard index of 1.5 at SRS down to 6.4 E-05 at RFETS. At SRS the hazard index was estimated to occur after about 735 years (the 10th consecutive 70-year lifetime); the maximum occurred both earlier and later at other sites. The dominant exposure pathway was groundwater ingestion at all sites except LANL, where it was resuspended soil ingestion. The key contaminants were carbon tetrachloride and mercury at all sites. This homogeneity in key contaminants is a result of the limited site-specific information available on hazardous chemical content of TRU waste.

Population The estimated radiological impacts to the surrounding population during the maximum 70-year lifetime over the 10,000-year analysis period ranged from 7.1 LCFs at RFETS down to 3.6 E-05 LCFs at ORNL. The time after loss of institutional control during which the maximum exposure occurred varied by site. At RFETS the 70-year lifetime period when the highest number of LCFs occurred was estimated to occur after about 665 years (the 8th consecutive 70-year lifetime period). The maximum numbers of LCFs occurred during the first lifetime at all other sites except SRS, where it occurred after about 9,905 years (141st lifetime period). The dominant exposure pathways was inhalation at all sites except Hanford, where it was resuspended soil ingestion. Transuranic radionuclides (Pu-238,-239,-240 and Am-241) were major contributors at all sites.

˙ The estimated hazardous chemical carcinogenic impacts to the surrounding population during the maximum 70-year lifetime over the 10,000-year analysis period were very small, much less than one cancer at every site. Impacts ranged from 2.8 E-04 cancers at RFETS down to 1.1 E-07 cancers at LLNL. At RFETS, the 70-year lifetime period when the highest fractional number of LCFs occurred was estimated to occur after about 525 years (the 7th consecutive 70-year lifetime period). The maximums occurred from 105 years to 665 years at the other sites. The dominant exposure pathways were resuspended soil ingestion and surface water ingestion. The key contaminant was 1,1,2,2-tetrachloroethane at all sites except LLNL, were they were beryllium and cadmium. Hazardous chemical noncarcinogenic impacts are not applicable for populations.

Aggregate impacts over 10,000 years The aggregate radiological impact from all seven sites totaled over the 10,000-year evaluation period was estimated to be 807 LCFs in the aggregate exposed population of 1.56 billion (the number of people exposed via air and groundwater-to-surfacewater

pathways around all seven sites over 10,000 years of 70-year lifetime periods). The majority of the impact occurred in the population around RFETS, which accounted for 781 LCFs. The aggregate impact consisted of 794 LCFs from release of the radionuclides in the No Action Alternative 2 inventory (Basic Inventory), plus 13 LCFs that would result from release of the "Additional Inventory." The "Additional Inventory" was included in the other action alternatives of SEIS-II, and consists of existing TRU waste including PCB-commingled TRU waste, commercial TRU waste, nondefense TRU waste, and mainly, TRU waste disposed of prior to the Atomic Energy Commission TRU waste policy of 1970. Most of this later category is estimated to be at DOE sites in shallow-land burial.

The aggregate hazardous chemical carcinogenic impact from all seven sites totaled over the 10,000-year evaluation period was estimated to be 0.002 cancers.

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This report analyzes two different types of human-health impacts. The first analysis addresses impacts of human intrusion at or near locations of former TRU waste storage following the loss of institutional control. Different intruder scenarios were examined for TRU waste assumed to be located in surface and shallow-buried (or other near-surface) storage facilities. Intruder scenarios may be characterized by the type of intrusion (active or inadvertent), initial location of the waste (surface or buried/near-surface), and type of exposure (acute or chronic). The following four scenarios were selected to examine the range of potential impacts that could result from human intrusion.

- Scavenger: active intrusion, surface waste, acute exposure
- Driller: active intrusion, buried waste, acute exposure
- Farmer: inadvertent intrusion, surface waste, chronic exposure
- Gardener: inadvertent intrusion, buried waste, chronic exposure.

The second type of analysis examined long-term, lifetime exposure of individuals and populations. No release of radionuclides or hazardous chemicals was assumed to occur until the end of the institutional control period. Maximally exposed individuals were assumed to be located at current site boundaries, and current off-site population distributions were used to estimate future impacts to surrounding populations. Impacts were evaluated over a 10,000-year period following loss of institutional control, for comparison to the performance assessment period of the WIPP repository under the SEIS-II action alternatives. Seventy-year lifetimes were assumed, and approximately 142 consecutive 70-year exposure periods were evaluated for the 10,000-year period. Individuals were assumed to be exposed via the groundwater-to-surface water pathways. This analysis included site-specific impacts at seven of the 10 major generator/storage sites: Hanford, INEEL, LANL, LLNL, ORNL, RFETS, and SRS. The analysis focused on these sites because 99% of the estimated TRU waste volume and inventory would be generated and stored there, and under the assumptions of this analysis, released from these sites as well.

Previous DOE NEPA analyses of the no action alternatives did evaluate the long-term impacts of leaving TRU waste at the generator/storage sites (DOE 1980; DOE 1990). Earlier analyses assumed that TRU waste would be in indefinite storage at major generator/storage sites. Institutional control, with effective monitoring and maintenance of storage facilities, would be sufficient to preclude any site impacts.

These earlier analyses referenced completed NEPA documents for some major retrievable storage facilities that describe the effects of continued retrievable storage. The sites include Hanford, INEEL, and SRS. The reader is referred to the published environmental documents for greater detail (DOE 1980, 1987, 1988).

For the analysis of long-term lifetime exposures, additional assumptions were made about the storage configurations of TRU waste at the sites, providing additional information on radionuclides and hazardous chemicals would be released. All waste was assumed to be placed into retrievable storage consistent with current practices at DOE sites. Current storage configurations include storage in soil-covered asphalt or concrete pads, shallow trenches, aboveground earthen berms, covered enclosures,

storage buildings, and buried caissons (RH-TRU waste only). The wastes would remain in the assumed storage configurations for a 100-year period of institutional control, which includes at its beginning a 10-year period of decommissioning of waste-handling and -treatment facilities. During this period, effective monitoring, surveillance, and maintenance of storage facilities was assumed sufficient to prevent the release of contaminants from these storage facilities.

1.2 Report Organization

This report provides supplemental information for the WIPP SEIS-II and describes the underlying bases, data, and assumptions used to estimate the impacts from postulated human intrusion and long-term, lifetime exposures to radionuclides and hazardous chemicals in TRU waste remaining at major generator/storage sites after loss of institutional control for No Action Alternative 2.

Section 2.0 provides detailed information on the analysis of impacts from human intrusion, including the methods used for radiological and non-radiological analysis, descriptions of the different intruder exposure scenarios, and the calculated impacts to individuals under these scenarios. Section 3.0 provides detailed information on the long-term, lifetime exposure analysis, including a description of the source's strength and configuration, scenarios considered, data and models used, and calculated impacts to individuals and populations near the sites. Section 4.0 provides references for the two analyses. Appendix A presents the radionuclide and hazardous chemical inventories used for both analyses in this report. Appendix B provides details on contaminant release and groundwater transport properties associated with the long-term, lifetime exposure analysis. Appendix C describes the geology and hydrology of the generator storage sites, and Appendix D describes the No Action Alternative 1A and 1B analysis.

2.2 Human Intrusion Exposure Scenarios

The human intrusion analyses evaluated the potential impacts to individuals at or near the location of TRU waste. The exposure scenarios were established to evaluate impacts from direct and inadvertent contact with the TRU waste, the differences between exposure to surface and shallow-buried waste, and the differences between acute and chronic exposures. Evaluations included exposures to radionuclide and hazardous chemicals for all scenarios. Estimation of impacts from human intrusion considered in this analysis included the following hypothetical scenarios: driller, gardener, scavenger, and farmer. The parameters and values employed for radiation dose and hazardous chemical exposures and associated impacts are presented in Table 2.1.

2.2.1 Direct Intrusion into Buried TRU Waste—Driller Scenario

With loss of institutional control, an inadvertent intruder could come into contact or become directly exposed to TRU waste located in shallow burial storage facilities by drilling into the waste. Under this driller scenario, an intruder was assumed to drill a water well directly through buried or soil-covered TRU waste. As a result of the drilling, contaminated soil was assumed to be brought to the surface and to mix with the topsoil. The circular drill hole through the TRU waste was assumed to be 30 cm in diameter and 4 m in depth. The contamination was assumed to be instantaneously mixed into the top 15 cm of clean surface soil. Further, it was assumed in this scenario that the extent of contamination was limited to an area of 10 m by 10 m.

The GENII calculations assumed that the driller intruder was exposed via external ground radiation for 40 hours, via inhalation of resuspended soil for 1 hour, and via inadvertent ingestion of soil for 5 days at a rate of 100 mg/day. The soil resuspension was based on an average mass loading factor of 1.0E-04 g_{soil}/ m^3_{air}. From these input parameters, a dose factor of dose per curie per cubic meter of soil down to 15 cm below surface was derived. The driller's breathing rate was assumed to be 20 m^3/24 hours for 40 hours. This was considered an acute scenario because of the limited exposure time of 40 hours.

2.2.2 Inadvertent Exposure to Buried TRU Waste—Gardener Scenario

Subsequent to waste being brought to the surface in the driller scenario, an individual was assumed to make a garden on the land, not knowing it contained the contaminated soil. The volume of contamination was limited to that of the drill tailings but was spread out over 2500 m^2. The individual was assumed to garden the plot over a period of 30 years. During this time, the gardener would receive 25% of his yearly vegetable and fruit intake from this garden. In addition to ingesting food crops, the individual was further exposed via inhalation of resuspended contamination, external radiation, and inadvertent ingestion of contaminated soil. The gardener was assumed to breathe all year at the location, 8,766 hours/year, but to be exposed externally only half the year (4,383 hours/year). This gardener scenario was considered chronic exposure because of the 30-year exposure duration.

Table 2.1. Parameters for Human Intrusion Scenarios

Exposure Time	Well Driller	30-year Gardener	24-hr Scavenger	30-year Farmer
Soil exposure time (h/y)	40	4383	24	4383
Inhalation exposure time (h/y)	1	8766	24	8766
Skin absorption exposure time (h/y)	40	8766	24	8766
Manual redistribution factor (m-1)	(a)	2.6E-04	(a)	(a)
Contaminated area (m^2)	100	2500	>2500[b]	>2500[b]
Cover soil mixing depth (cm)	15	15	0	0
Mass loading (g/m^3)	1.0E-04	1.0E-04	1.0E-4	1.0E-4
Soil ingestion exposure time (d/y)	5	365	1	365
Soil ingestion rate (mg/d)	100	100	100	100
Breathing rate (m^3/d)	20	20	20	20
Ingestion pathway parameters				
Fraction of diet from local (contaminated) region				
leafy, fruit & other vegetables		0.25		1.0
grain		(a)		1.0
Growing time				
leafy vegetables		45		45
other vegetables		90		90
fruit		90		90
grains		(a)		90
Consumption rate (kg/y)				
leafy vegetables		3.5		14
other vegetables	(a)	14	(a)	55
fruit		8		31
grains		(a)		73
Animal Products				
beef				42
poultry				11
cow's milk		(a)		110
eggs				10

(a) Not an exposure route.
(b) External pathway not limited by contaminated area extent.

2.2.3 Direct Intrusion into Surface-Stored Waste—Scavenger Scenario

With loss of institutional control, inadvertent intruders may come into more direct contact and be directly exposed to waste in surface storage facilities. In this scenario, a scavenger was assumed to come into direct contact with surface-stored TRU waste over a 24-hour period. The scavenger was exposed by inhalation of resuspended soil, external radiation, and inadvertent ingestion of contaminated soil while at the site. The scavenger was assumed to breathe 20 m³/day. It also was assumed that no clean soil covers the waste, so the dose factors per unit concentration were multiplied by the waste-form concentration to get the total dose to the scavenger. This scavenger scenario was considered to be an acute exposure scenario because of the 24-hour exposure.

2.2.4 Inadvertent Intrusion of Surface-Stored Waste—Farmer Scenario

Another potential intruder scenario for near surface waste examined was a farmer who was assumed to live and farm on a plot of land immediately over the near surface waste once the waste degrades to the point where it becomes indistinguishable from the surrounding land. For this scenario, a farmer was assumed to work the land for a 30-year period, getting 100% of his produce and animal products from the farm. The adult farmer was exposed via ingestion of contaminated food crops and animal products, inhalation of resuspended contamination, external radiation, and inadvertent ingestion of contaminated soil. The area of contamination that the farmer works is equal to the areal footprint of the TRU waste. Because no surface soil was assumed, the TRU concentration of the farmed land was equal to the concentration in the TRU waste. The farmer also was assumed to live year round (8,766 hours/year) at the waste site, but was exposed externally only half the time (4,383 hours/year). This farmer scenario was considered a chronic exposure scenario.

The difference between the farmer and gardener scenarios, was that for the farmer scenario, the area of contamination was larger, the concentration of waste was greater, there was no surface cover, and the farmer's ingestion pathway was greater, and included grains as well as animal products.

2.3 Impacts of Human Intrusion

This section presents the results of the human intrusion analysis conducted for the WIPP SEIS-II No Action Alternative 2. The section is divided into radiological impacts and impacts from hazardous chemical exposure for four intruder exposure scenarios representing exposures from buried and surface TRU waste, direct and inadvertent intrusion, and acute and chronic exposures. Site-specific impacts for seven major generator/storage sites are presented.

2.3.1 Radiological Impacts

Radiological impacts are presented below for the driller, gardener, scavenger, and farmer scenarios for each of the seven major generator/storage sites. Results are presented for exposure to CH-TRU waste at all sites and for separate exposure to RH-TRU waste at Hanford, INEEL, LANL, and ORNL. LLNL, RFETS,

and SRS did not have RH-TRU waste in the SEIS-II Basic Inventory. Radiation dose estimates are *presented* in Table 2.2, and radiological impact estimates, as the probability of an LCF in an exposed individual, are presented in Table 2.3.

Driller Scenario

The estimated probability of an LCF to a hypothetical driller from exposure to CH-TRU waste ranges from 1.1E-06 to 5.4E-06 for the seven sites. The estimated probability of an LCF to the driller from exposure to RH-TRU waste ranges from 1.1E-06 to 2.9E-05 for the four sites with RH-TRU waste. The highest estimated probability of an LCF for exposure to CH-TRU waste was at the INEEL. Americium-241 was the radionuclide contributing 97% to the dose via the external pathway. The highest estimated probability of an LCF for exposure to RH-TRU waste was at Hanford. Cesium-137 was the radionuclide contributing 89% to the dose via the external pathway.

Table 2.2. Radiation Dose from CH-TRU and RH-TRU for Intruder Scenarios by Site

Intruder Scenario	Hanford	INEEL	LNLL	LANL	ORNL	RFETS	SRS
CH-TRU **(Dose, in Rem)**							
Driller	2.2E-03	0.011	(a)	4.5E-03	0.010	(a)	7.3E-03
Gardener/First Year	0.81	1.1		1.7	1.3		5.5
Gardener/30-Year	19	26		41	29		126
Scavenger	3.1	4.2	1.3	6.6	5.2	38	21
Farmer/First Year	1,00	1,600	475	2,400	1,900	14,000	7,700
Farmer/30-Year	26,000	36,000	11,000	55,000	43,000	320,000	170,000
RH-TRU **(Dose, in Rem)**							
Driller	89	17	(b)	6.1	14	(b)	(b)
Gardener/First Year	4.3	0.86		0.28	0.70		
Gardener/30-Year	89	17		6.1	14		
Scavenger	24	4.1		1.4	2.9		
Farmer/First Year	10,000	2,700		550	2,700		
Farmer/30-Year	220,000	57,000		12,000	55,000		
(a) Not applicable because site does not have buried TRU waste.							
(b) Not applicable because site does not have RH-TRU waste.							

Table 2.3. Radiological Impacts from CH-TRU and RH-TRU for Intruder Scenarios by Site

Intruder Scenario	Hanford	INEEL	LNLL	LANL	ORNL	RFETS	SRS
Radiological Impacts from CH-TRU (Probability of an LCF)							
Driller	1.1E-06	5.4E-06	(a)	2.3E-06	5.2E-06	(a)	3.7E-06
Gardener/First Year	4.1E-04	5.6E-04		8.6E-04	6.3E-04		2.7E-03
Gardener/30-Year	9.6E-03	0.013		0.021	0.015		0.063
Scavenger	1.6E-03	2.1E-03	6.4E-04	3.3E-03	2.6E-03	0.019	0.010
Farmer/First Year	0.56	0.79	0.24	1.0	0.95	1.0[b]	1.0[b]
Farmer/30-Year	1.0[b]	1.0[b]	1.0[b]	1.0[b]	1.0[b]	1.0[b]	1.0[b]
Radiological Impacts from RH-TRU (Probability of an LCF)							
Driller	2.9E-05	4.9E-06	(c)	1.1E-06	3.5E-06	(c)	(c)
Gardener/First Year	2.2E-03	4.3E-03		1.4E-04	3.5E-04		
Gardener/30-Year	0.045	8.7E-03		3.6E-03	7.0E-03		
Scavenger	0.012	2.1E-03		6.9E-04	1.4E-03		
Farmer/First Year	1.0[b]	1.0[b]		0.27	1.0[b]		
Farmer/30-Year	1.0[b]	1.0[b]		1.0[b]	1.0[b]		

(a) Not applicable because site does not have buried TRU waste.
(b) Value is limited to 1.0, calculated value is higher.
(c) Not applicable because site does not have RH-TRU waste.

Gardener Scenario

The estimated first-year probability of an LCF for the gardener exposed to CH-TRU waste, presented in Table 2.3, ranges from 1.7E-04 to 5.0E-03, while the 30-year probability of an LCF ranges from 9.6E-03 to 0.063. The estimated first-year probability of an LCF to the gardener from exposure to RH-TRU waste ranges from 1.4E-04 to 2.2E-03, while the 30-year probability of an LCF ranges from 3.6E-03 to 0.012. The highest estimated first-year probability of an LCF from exposure to CH-TRU waste was at the SRS with Pu-238 contributing 91% of the dose by the inhalation pathway. The highest estimated first year probability of an LCF from exposure to RH-TRU waste was at Hanford where Cs-137, Sr-90, Am-241, and Pu-239 together contributed 96% of the dose. The external pathway contributed 62% to the radiation dose, the rest was split approximately evenly between inhalation and ingestion pathways.

Scavenger Scenario

The estimated maximum probability of an LCF to a scavenger from exposure to CH-TRU waste ranged from 6.4E-04 to 0.019. The highest impacts for CH-TRU waste were estimated at RFETS with Am-241 and Pu-239 contributing 87% of the dose by the inhalation pathway. The estimated probability of an LCF to the scavenger from RH-TRU waste ranges from 6.9E-04 to 0.012. The highest probability of impacts for RH-TRU waste were estimated for Hanford, where Cs-137 contributed 88% of the dose by the external pathway.

Farmer Scenario

Of all the intruder scenarios, estimated impacts for the farm intruder scenario were substantially higher than any of the other scenarios. This was because the farm intruder scenario had the most exposure routes and exposure time of all the scenarios, compounded with undiluted waste concentrations. In fact, in the first few years' exposure, health impacts for the farm intruder scenario for all sites were estimated to be highest possible (i.e., probability of cancer equal to 1.0). The health impacts between CH- and RH-TRU waste were generally equivalent, although there were fewer sites that contain RH-TRU waste. The estimated first-year probability of an LCF ranged from 0.24 to 1.0 for CH-TRU waste and 0.27 to 1.0 for RH-TRU waste. The RFETS had the highest estimated health impacts for the CH-TRU waste farm intruder scenario (1.0 probability of an LCF in the first year). The inhalation pathway accounted for 78% of the RFETS dose, while ingestion accounted for 21%. The radionuclides contributing the greatest were Am-241, Pu-239, and Pu-240 with 58%, 29%, and 12% of the dose, respectively. For the RH-TRU waste, the Hanford Site had the highest estimated health impacts for all scenarios. The highest estimated first-year probability of an LCF from exposure to RH-TRU waste was at Hanford where Sr-90 and Cs-137 together contributed 88% of the dose. The ingestion pathway contributed 62%, and the external pathway contributed 32% to the first year's dose. For the 30-year exposure farmer scenario, the estimated probability of an LCF for all sites was 1.0. Noncarcinogenic radiation impacts such as radiation pneumonitis in the lungs would likely occur.

2.3.2 Non-Radiological Impacts

Non-radiological impacts from exposure to heavy metals and organic chemicals are presented below for the driller, gardener, scavenger, and farmer scenarios. Impacts are presented as both the probability of cancer incidence from exposure to carcinogens and the potential for reaching the threshold for noncarcinogenic health effects from exposure to non-carcinogens. As noted in Section 2.2, site-specific hazardous chemical inventories were not available, so no site-specific calculations were made. In general, impacts from exposures to heavy metal in the intruder scenarios were estimated to be substantial, while impacts from organic chemical exposure were much lower. Table 2.4 presents the results for exposure to heavy metals, while impacts from organic chemical exposure are presented in Tables 2.5 and 2.6 for CH-TRU and RH-TRU waste, respectively.

Table 2.4. Impacts from Heavy Metal Exposures for All Intruder Scenarios[a]

Scenario	Probability of Cancer Incidence [b]			
	Beryllium	Cadmium	Lead	Mercury
Driller	1.3E-07	1.4E-09	(c)	(c)
Gardener (30-year)	1.4E-03	1.5E-05		
Scavenger	1.7E-04	1.8E-06		
Farmer (30-year)	1.0[d]	0.02		

Scenario	Noncarcinogenic Impact[b] HI[e] or Fraction of PEL[f]			
	Beryllium	Cadmium	Lead	Mercury
Driller (PEL)	17	0.098	CH: 27 RH: 3000	12
Gardener (First Year) (HI)	0.078	0.11	CH: 36 RH: 3900	77
Scavenger (PEL)	91	5.2	CH: 1,400 RH:160,000	620
Farmer (First Year) (HI)	100	15	CH:48,000 RH:5,200,000	100,000

(a) Impacts from exposure to hazardous chemicals were assumed to be the same at all sites.
(b) Results were independent of whether the waste was CH-TRU or RH-TRU except where noted.
(c) Contaminant does not have known carcinogenic effects.
(d) Value is limited to 1.0, calculated value is higher.
(e) Hazard index of greater than 1.0 would predict the occurrence of the chemical specific noncarcinogenic health effect.
(f) PEL is the OSHA permissible exposure level averaged over an 8-hour work day.

Driller Scenario

For the driller scenario, the estimated probability of cancer incidence from exposure to heavy metals was 1.3E-7 or less. Chemical noncarcinogens in the CH-TRU driller scenario were in general 10 to 30 times above the PEL allowed in short-term occupational settings. However, in the RH-TRU waste, lead was 3,000 times the PEL at all sites. No health effects would be expected from exposure to organic chemicals at any site in the driller scenario. The maximum cancer incidence probability is 7E-9 from exposure to 1,1,2,2-tetrachloroethane in CH-TRU waste, and the maximum fraction of PEL concentration is 2E-3 for 1,1,2,2-tetrachloroethane for both CH- and RH-TRU waste.

Gardener Scenario

The estimated probability of cancer incidence from exposure to chemical carcinogens in the gardener intruder scenario for beryllium and cadmium were estimated to be 1.4E-03 and 1.5E-05, respectively. For the CH-TRU chemical noncarcinogens, lead and mercury were estimated to have an HI of 36 and 77,

Table 2.5. Impacts from CH-TRU Organic Chemical Exposures for all Intruder Scenarios[a]

Hazardous Chemical	Probability of Cancer Incidence [b]			
	Driller	Gardener (30-year)	Scavenger	Farmer (30-year)
Carbon Tetrachloride	2E-11	2E-07	2E-08	2E-04
Chlorobenzene	NA[b]			
Chloroform	4E-12	4E-08	5E-9	5E-05
Methyl Ethyl Ketone	NA[b]			
Methylene Chloride	3E-12	3E-08	3E-09	4E-05
1,1,2,2-Tetrachloroethane	7E-09	8E-05	9E-06	0.1
Toluene	NA[b]			
1,1-Dichloroethylene	2E-12	2E-08	2E-09	2E-05
1,2-Dichloroethane	5E-12	5E-08	7E-09	7E-05

Hazardous Chemical	Noncarcinogenic Impact HI[c] or Fraction of PEL[d]			
	Driller (PEL)	Gardener (First Year) (HI)	Scavenger (PEL)	Farmer (First Year) (HI)
Carbon Tetrachloride	5E-05	0.01	3E-03	15
Chlorobenzene	3E-06	2E-3	2E-04	3
Chloroform	1E-05	1E-04	5E-04	0.2
Methyl Ethyl Ketone	1E-06	3E-05	6E-05	0.03
Methylene Chloride	2E-06	7E-04	1E-04	0.9
1,1,2,2-Tetrachloroethane	2E-03	NA[e]	0.1	NA[e]
Toluene	7E-07	3E-05	4E-05	0.04
1,1-Dichloroethylene	NA[f]	2E-05	NA[f]	0.03
1,2-Dichloroethane	6E-07	NA[e]	3E-05	NA[e]

(a) Impacts from exposure to hazardous chemicals were assumed to be the same at all sites.
(b) Contaminant does not have known carcinogenic effects.
(c) Hazard index of greater than 1.0 would predict the occurrence of the chemical-specific noncarcinogenic health effect.
(d) PEL is the OSHA permissible exposure level averaged over an 8-hour work day.
(e) Contaminant does not have a reference dose.
(f) Contaminant does not have a PEL.
NA = Not applicable.

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Table 2.6. Impacts from RH-TRU Organic Chemical Exposures for all Intruder Scenarios[a]

Hazardous Chemical	Probability of Cancer Incidence			
	Driller	Gardener (30-year)	Scavenger	Farmer (30-year)
Carbon Tetrachloride	7E-12	7.8E-08	9E-09	1.0E-04
Chlorobenzene	NA[b]			
Chloroform	3E-12	3.4E-08	4E-09	4.5E-05
Methyl Ethyl Ketone	NA[b]			
Methylene Chloride	4E-12	4.4E-08	5E-09	5.9E-05
1,1,2,2-Tetrachloroethane	6E-09	6.1E-05	7E-06	0.081
Toluene	NA[b]			
1,1-Dichloroethylene	2E-12	1.7E-08	2E-09	2.2E-05
1,2-Dichloroethane	6E-12	6.4E-08	8E-09	8.5E-05
	Noncarcinogenic Impact HI[c] or Fraction of PEL[d]			
Hazardous Chemical	Driller (PEL)	Gardener (First Year) (HI)	Scavenger (PEL)	Farmer (First Year) (HI)
Carbon Tetrachloride	2E-05	4.9E-03	1E-03	6.5
Chlorobenzene	4E-06	2.8E-03	2E-04	3.7
Chloroform	9E-06	9.9E-05	5E-04	0.13
Methyl Ethyl Ketone	1E-06	2.9E-05	6E-05	0.038
Methylene Chloride	3E-06	1.1E-03	2E-04	1.4
1,1,2,2-Tetrachloroethane	2E-03	NA[e]	0.09	NA[e]
Toluene	9E-07	3.4E-05	5E-05	0.045
1,1-Dichloroethylene	NA[f]	2.4E-05	NA[f]	0.032
1,2-Dichloroethane	7E-07	NA[e]	4E-05	NA[e]

(a)   Impacts from exposure to hazardous chemicals were assumed to be the same at all sites.
(b)   Contaminant does not have known carcinogenic effects.
(c)   Hazard index of greater than 1.0 would predict the occurrence of the chemical-specific noncarcinogenic health effect.
(d)   PEL is the OSHA permissible exposure level averaged over an 8-hour work day.
(e)   Contaminant does not have a reference dose.
(f)   Contaminant does not have a PEL.
NA = Not applicable.

respectively. The lead HI was 3,900 for RH-TRU waste, and all other HIs were below 1.0. An HI of 1.0 or greater would indicate that the intake exceeds the reference dose, and that effect would be expected in the affected individual. No health effects would be expected from exposure to organic chemicals at any site in the gardener scenario. The maximum cancer incidence probability is 8E-5 from exposure to 1,1,2,2-tetrachloroethane, and the maximum first year HI is 0.01 from exposure to carbon tetrachloride in CH-TRU waste.

Scavenger Scenario

In the scavenger scenario, the probability of cancer incidence does not exceed 1.7E-4. However, the air concentrations of hazardous metals were estimated to be substantially above the PELs for all the hazardous metals. The concentrations were estimated to be 5.2, 91, 620 and 1,400 times the PEL for cadmium, beryllium, mercury, and lead in CH-TRU and up to 160,000 times the PEL for lead in RH-TRU. No health effects would be expected from exposure to organic chemicals at any site in the scavenger scenario. The maximum cancer incidence probability is 9E-6 from exposure to 1,1,2,2-tetrachloroethane in CH-TRU waste, and the maximum fraction of PEL concentration is 0.1 for 1,1,2,2-tetrachloroethane for CH-TRU waste.

Farmer Scenario

For the farmer scenario, the impacts were the highest. The carcinogenic impacts included a 1.0 probability of cancer incidence from exposure to beryllium in TRU waste. The carcinogenic impact from cadmium was a probability of cancer incidence of 0.02. The noncarcinogenic HI impacts were 15, 100, 48,000 and 100,000 for cadmium, beryllium, lead, and mercury in CH-TRU waste and were estimated to be up to 5,200,000 for lead in RH-TRU waste. Assumptions were made that were very conservative in this and the other scenarios. First, it was assumed that, however unlikely, someone would try to farm directly on TRU waste. Second, it was assumed that all the heavy metals were in a powder form and were available for resuspension and respiration. These two assumptions produced substantially high estimated impacts. Calculated impacts could be lessened by adjusting these assumptions.

Potential impacts from CH-TRU were greater than those from RH-TRU. The probability of cancer incidence was 0.1 for the farmer from exposure to 1,1,2,2-tetrachloroethane in CH-TRU waste. The maximum HI was 15 for carbon tetrachloride. Therefore, some health effects would be expected from the farmer's exposure to organic chemicals.

3.0 Long-Term, Lifetime Exposure Analysis

This chapter describes the analysis of potential impacts from long-term, lifetime exposure to radionuclides and hazardous chemicals in TRU waste under the SEIS-II No Action Alternative 2 after loss of institutional control at major generator/storage sites. Earlier NEPA analyses of the no action alternative in the FEIS (DOE 1980) and SEIS-I (DOE 1990) did not address long-term environmental and human-health impacts. The assumptions in these earlier analyses were that TRU wastes would be in indefinite storage at generator/storage sites and that institutional control, with effective monitoring and maintenance of storage facilities, would be sufficient to preclude any site impacts. These earlier analyses focused on expected site operations associated with treatment and storage.

Under the SEIS-II No Action Alternative 2, TRU waste would not be emplaced at the WIPP. TRU waste would be consolidated at 10 major generator/storage sites during a 35-year waste generation period with the waste placed in retrievable storage consistent with current practices. Current storage configurations include surface and buried storage: soil-covered asphalt or concrete pads, shallow trenches, aboveground earthen berms, covered enclosures, storage buildings, and, for RH-TRU waste only, buried caissons. The waste would remain in the assumed storage configurations for a 100-year period of institutional control. During this period, effective monitoring, surveillance, and maintenance of storage facilities was assumed sufficient to prevent the release of contaminants. At the end of the 100-year period, institutional control was assumed to be lost and stored waste and associated facilities begin to degrade, allowing waste to be released to the accessible environment. Long-term, lifetime exposure analysis of impacts began at this point; no impacts were evaluated during the institutional control period.

For the long-term, lifetime exposure analysis, human-health impacts were evaluated for 10,000 years following the loss of institutional control, on the basis of 70-year lifetimes for individuals and 70-year periods for populations. Impacts were estimated at seven of the 10 major generator/storage sites: Hanford, INEEL, LANL, LLNL, ORNL, RFETS, and SRS. The analysis focused on these seven major sites because 99% of the estimated TRU waste volume and inventory would be generated/stored there. Other sites considered but not examined in this analysis include ANL, Mound, and NTS. Transport through air and groundwater were considered for exposure of individuals and population. Individuals were assumed to be exposed directly through air and groundwater pathways. Populations were exposed via the atmospheric and surfacewater pathways, with groundwater providing the transport of contaminants to surfacewater. Groundwater was not considered as a direct exposure pathway for populations because it is not a substantial water source for the populations considered.

This chapter describes the methodology and models used for the long-term, lifetime exposure analysis. The TRU waste generator/storage sites and site-specific inventories used are described next, and then the waste configuration and release scenarios are defined. The transport and exposure scenarios used are also described, including the maximally exposed individual (MEI) and populations potentially exposed at each waste generator/storage waste site. Finally, the estimated impacts from long-term, lifetime exposures to radionuclides and hazardous chemicals are presented.

3.1 Methodology

Potential human health impacts from exposure to radionuclides and hazardous chemicals were considered in this analysis. Impacts from exposure to ionizing radiation from radionuclides was evaluated for the potential to cause latent cancer fatalities (LCFs). Impacts from exposure to hazardous chemicals were evaluated for the potential to cause cancer and noncarcinogenic health effects. These three types of impacts are directly related to the three types of unit health impact factors (UIFs) computed for this study. To facilitate such an effort, the modular risk analysis (MRA) methodology was used as described below.

3.1.1 Modular Risk Analysis Methodology

The MRA methodology was developed by Pacific Northwest National Laboratory and Advanced Sciences, Inc., to facilitate regional-scale risk analysis. This methodology is described in several documents (Strenge and Chamberlain 1995; Whelan et al. 1995) and presentations (Whelan et al. 1994). The MRA methodology was developed for regional and site-wide risk computations involving a large number of release sites and contaminants with different waste forms and for various environmental settings and transport and exposure pathways.

The MRA methodology is based on the assumptions of linearity between the release site source, environmental transport, and impacts at the receptor. By determining the linearity of the system, the methodology can be divided into compartments that can be implemented both independently and concurrently. The compartments of the MRA methodology are 1) contaminant mass at the source, 2) determination of contaminant-release rate from the source, 3) transport modeling of the contaminant into the environment (environmental concentrations at the receptor location), 4) exposure assessment for dose to receptor (maximally exposed individual or population), and 5) estimation of risk at the receptor.

The MRA methodology is based on the following general description for risk/hazard:

$$\text{Health Impact} = P \times RF \times URF \tag{3.1}$$

where health impact = estimated probability of adverse effects (carcinogenic risk for radionuclides and chemical carcinogens and hazard quotient for noncarcinogenic constituents) from a contaminant at a receptor.

P = probability of the release event (unitless).

RF = releasable fraction of the source (unitless).

URF = health impact associated with a contaminant at a receptor, based on a unit quantity at the source and has a unit of health impact per source mass. (NOTE: Source mass is often expressed as a source activity for radioactive contaminants.)

3.2

Therefore, the URF, is determined as follows:

$$URF = [(Q \times UFF) \bullet UTF] \times UDF \times UIF \tag{3.2}$$

where Q = estimated mass of contaminant at the source in grams or curies

UFF = time series of contaminant-release-rate fluxes designated as contaminant mass per time divided by unit contaminant mass.

UTF = time series of environmental concentrations at a receptor produced from the UFF for groundwater, air, surfacewater, and soil media (expressed as contaminant mass per volume of medium divided by unit contaminant mass per time).

UDF = dose to an organism from a unit concentration for a given exposure pathway. For chemicals, UDF is expressed as contaminant mass per body mass per time divided by unit contaminant mass per volume of contaminant in the environment at the receptor point. For radionuclides, UDF is expressed as contaminant total dose (rem) divided by unit contaminant mass per volume of contaminant in the environment at the receptor point.

UIF = unit health-impact factor that provides the dose-conversion factor for radionuclides, cancer potency factor for chemical carcinogens, or reference dose for noncarcinogenic contaminants. For radionuclides, UIF is expressed as cancer fatalities divided by unit contaminant total dose. For chemical carcinogens, UIF is expressed as cancer incidence divided by unit contaminant mass per body mass per time. For chemical noncarcinogens, UIF is expressed as hazard quotient divided by unit contaminant mass per body mass per time.

The UTF and UFF require the convolution of time series, and those products can be combined with the UDF by straight multiplication. Whereas Equation (3.2) describes the link between the different unit factors involved in computing the URF, Equations (3.3) through (3.6) describe each unit factor that was developed to compute URFs. Note that the UFF and UTF are time series and need to be convoluted together. The source, UDF, and UIF are multipliers.

$$UFF = \frac{F_s}{S_u} \tag{3.3}$$

where F_s = contaminant-flux-release rate from the TRU waste form in grams per year or curies per year.

S_u = unit source mass in grams or source activity in curies.

The UFF includes the probability of release and the release-factor fraction for a given scenario.

$$UTF = \frac{C_f}{F_u} \tag{3.4}$$

3.3

where C_r = contaminant concentration at the receptor, based on transport through the appropriate medium in grams per cubic meter or curies per cubic meter divided by grams per year or curies per year (air), grams per square meter or curies per square meter divided by grams per year or curies per year (soil), or grams per milliliter or curies per milliliter divided by grams per year or curies per year (water).

F_u = unit contaminant flux in grams per year or curies per year.

$$UDF = \frac{D_c}{C_u} \tag{3.5}$$

where D_c = unit dose from the contaminant in grams of contaminant per kilogram of body mass per day or total dose (rem) to a human at the receptor.

C_u = unit concentration at the receptor in grams per cubic meter or curies per cubic meter divided by grams per year or curies per year (air), grams per square meter or curies per square meter divided by grams per year or curies per year (soil), or grams per milliliter or curies per milliliter divided by grams per year or curies per year (water), based on contaminant transport through the appropriate medium.

$$UIF = \frac{R_d}{D_u} \tag{3.6}$$

where R_d = risk associated with a unit dose as risk per gram of contaminant per kilogram of body mass per day or risk per total dose.

D_u = unit dose to a human in grams of contaminant per kilogram of body mass per day or total dose (rem).

The URFs were developed for the different environmental settings of the seven major generator/ storage sites. Environmental settings were assumed to have homogeneous climatology, hydrology, and geologic characteristics (see Section 3.2 and Appendix C). Therefore, the URF is representative of the impacts from a release site within a region, not actual impacts. The local climatology, hydrology and geologic characteristics for this analysis were developed and published in Holdren et al. (1995).

Conceptual site models were developed for each environmental setting associated with a storage site. These conceptual site models define the relationship between the source contaminant at the release sites and the health impacts at the receptors. The important components associated with these relationships are the contaminants of interest, waste-source types and forms, release mechanisms, exposure media, and receptor types. For this analysis, the probability of a release or exposure event is assumed to be 1.0. Likewise, it is assumed that sources are in a releasable form such that the releasable fraction of the source is equal to 1.0.

Once the waste configuration and TRU waste forms for each environmental setting have been identified, the release mechanisms need to be selected. For this analysis, infiltration of contaminants to groundwater and surfacewater were considered. Suspension, volatilization, and overland-flow release

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mechanisms also were considered. The resulting release rate (contaminant flux) for each release mechanism also depends on the TRU waste form. In addition, the solubility of each contaminant in the TRU waste form has potential for being an important factor in determining contaminant-release rates. This analysis considered two different TRU waste forms: 1) soil/debris, and 2) cement. An analysis of the solubility limits for the primary TRU waste contaminants at each site was conducted (see Appendix B), and results were incorporated into computations of contaminant fluxes.

The UFFs were generated using a computer model (Streile et al. 1996) to simulate the release of contaminants from a source term. The model directly considers contaminated soil/debris and solidified (cemented) TRU waste forms. The contaminant is removed from the source by simultaneously evaluating degradation or decay, groundwater leaching, atmospheric volatilization, and soil erosion by wind suspension and overland runoff, as appropriate. Because of potential solubility issues associated with the TRU waste forms analyzed, the contaminant mass (Q) was included with the UFF to produce total flux factors. These were subsequently convoluted with the UTFs, thereby resolving the solubility issue.

The UTF represents the environmental fate and transport component of the unit factor methodology. The UTF value is based on 1 g (1 Ci) of contaminant at the source per year, which, after being transported through a specific environmental medium, ultimately arrives at the receptor exposure point. An important transport parameter linked to the exposure points is the transport distance from the source to the receptor. The transport distance for groundwater analysis is a well 300 m directly downgradient from the source, assumed centerline of the plume. The atmospheric transport distance is at least 100 m away from but within an 80-km radius of the release point. The UTF is media dependent.

UIF relates the chemical intake or radiation dose to a risk or hazard index, as appropriate, for each contaminant. Both UIF and UDF are defined for intake or exposure routes of inhalation, ingestion, and external radiation.

Finally, both MEI and population endpoints were considered. MEI endpoints influence UDF calculations by defining dose intake factors and UTF calculations by defining the exact location of the receptor. The combination of these factors requires the convolution of time series and straight multiplication. The UFF and UTF are time series at different locations. The UFF is the time series of contaminant-release rate from the source, while the UTF is the time series of contaminant concentration at the receptor point. Equation (3.7) provides the convolution method used to combine the series.

$$UTF = [Q \times UFF] \bullet UTF(\tau) = \int_0^\tau [Q \times UFF(t)] \times UTF(\tau-t)dt \qquad (3.7)$$

Once the convolution is completed, all the factors can be combined based on Equation (3.2) to provide health risk or hazard quotient impact values.

The units for air UTFs are either 1) grams of contaminant per cubic meter of air divided by gram per year of contaminant at the source, or 2) curies of contaminant per cubic meter of air divided by curie per year of contaminant from the source. The units for soil are either 1) grams of contaminant

per square meter of soil divided by gram per year of contaminant from the source, or 2) curies of contaminant per square meter of soil divided by curie per year of contaminant from the source. The units for surfacewater or groundwater are either 1) grams of contaminant per milliliter of water divided by gram per year of contaminant from the source, or 2) curies of contaminant divided by milliliter of water curie per year of contaminant from the source.

The UDF involves an average daily intake in grams of contaminant per kilogram of body mass per day for chemicals or a lifetime radiation dose in rem for radionuclides. The calculations for population risk were computed in two basic media, surfacewater and air. Population UDF and UIF values are calculated according to Equations (3.5) and (3.6) for individual UDF and UIF values. The difference being that the population UEF risk values in the output files must be taken from files generated specifically for population exposures. The specific population data used for this analysis is derived from site-specific information. The UDF values differ from UIF values in that the population UDF is evaluated using average parameter values instead of 90th percentile values used for MEI impacts. There are no population UDF and UIF for noncarcinogenic chemicals.

For atmospheric transport, the same approach is inappropriate because the distances and directions between individuals can vary greatly. The varying distances imply that concentrations to which different individuals are exposed also can vary greatly. To adjust for differences in concentration, an effective atmospheric population value is used. The effective atmospheric population in an angular segment is equal to the population in the angular segment times the concentration in the same angular segment divided by the peak concentration. The peak concentration is the concentration used in the MEI calculation. The total effective population is the summation of the effective population for each angular segment. Because dispersion is unique to each contaminant, the effective population was computed for each contaminant. To compute the atmospheric population risk, the effective population is multiplied by the MEI risk. Equation (3.8) shows how the effective atmospheric population is computed.

$$EAP_j = \sum_{AS}^{i=1} \frac{P_i \, C_{i,j}}{PC_j} \qquad (3.8)$$

where EAP_j = effective atmospheric population for contaminant j (number of people).
 AS = angular segment (160 based on 16 directions and 10 distances [unitless]).
 P_i = population assigned to angular segment I (number of people).
 $C_{i,j}$ = contaminant j atmospheric concentration for angular segment I (grams or curies per cubic meter).
 PC_j = peak contaminant j atmospheric concentration (gram or curies per cubic meter).

3.1.2 MEPAS Computer Code

The potential health impacts from exposure to radionuclides and hazardous chemicals were evaluated with a DOE-developed computer code. The Multimedia Environmental Pollutant Assessment System (MEPAS) code was used to calculate contaminant fluxes from the source, environmental fate and transport to the receptor point, and toxicological impacts and carcinogenic risks from hazardous

chemicals and radionuclides (Droppo et al. 1989a, 1989b; Whelan et al. 1987; Strenge and Peterson 1989; Buck et al. 1995; Streile et al. 1996). A brief discussion of the key components of these codes used in this analysis follows.

The MEPAS code integrates and evaluates contaminant release, transport, and exposure pathways for chemicals and radioactive releases according to their potential human-health impacts. MEPAS is a physics-based approach that couples contaminant release, migration, and fate for environmental media with exposure routes and chronic health consequences for radionuclides and hazardous chemicals (carcinogenic and noncarcinogenic effects). Figure 3.1 shows a simplified diagram of how MEPAS works.

Contaminant release from the waste zone was modeled by MEPAS source-term model (Streile et al. 1996; Buck et al. 1995). In general, the mass or activity of a contaminant in the source zone decreases over time because of contaminant removal by first-order degradation or radioactive decay, leaching to the groundwater, wind suspension, surfacewater erosion, and volatilization.

Hazardous airborne and waterborne radioactive and hazardous contaminant transport in multiple media was evaluated by the MEPAS atmospheric model (Whelan et al. 1987; Droppo et al. 1989a, 1989b). The MEPAS atmospheric model considers the input of suspension and volatilization-release rates to compute transport and dilution, washout by cloud droplets and precipitation, and deposition on the underlying surface cover. The atmospheric model uses climatological information on wind speed
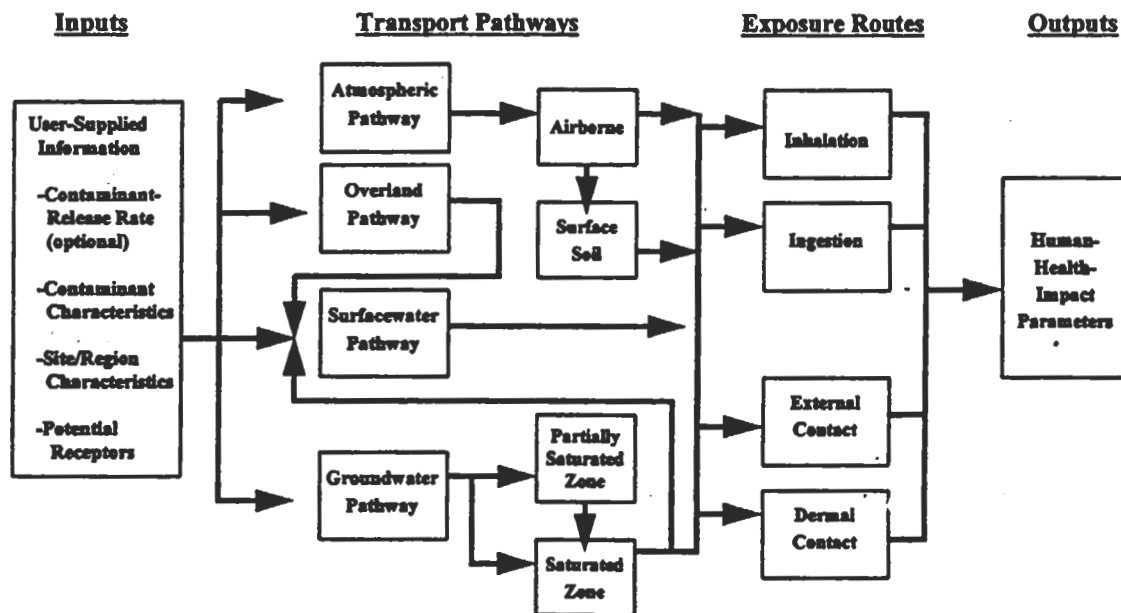
Figure 3.1. Simplified Diagram Outlining Various Linkages of MEPAS

and direction, precipitation, and atmospheric stability to compute average air and surface contamination concentrations. Contaminant transport is assumed to occur sufficiently fast so that chemical transformation can be neglected.

The MEPAS waterborne transport model (Whelan et al. 1987; Droppo et al. 1989a, 1989b) consists of two main models: 1) groundwater, and 2) surfacewater. These transport models can be run either separately or linked to provide environmental concentrations at specified receptor points. For each waterborne transport pathway, contaminant retardation is described by an equilibrium coefficient, K_d. First-order degradation or decay is assumed for all contaminants that do not result in toxic decay products. On reaching the environmental receptor point, radiological decay is corrected using the Bateman equation, and the model subsequently computes the temporal distribution of each decay progeny.

Results from the different transport pathway models are used as input to the exposure model to calculate the human-health impacts for each contaminant. The following exposure routes are considered to determine the potential exposure of individuals and the surrounding population: 1) dermal contact (not used in this analysis), 2) external exposure from radionuclides, 3) inhalation, and 4) ingestion. Each exposure route is evaluated to obtain an estimated average daily human exposure from each contaminant. The daily exposure rates are then converted, using mathematical models, to average individual risk factors for radionuclides, carcinogenic chemicals, and noncarcinogenic chemicals.

3.2 TRU Waste at Generator/Storage Sites

The following subsections describe the regional settings, TRU waste-storage facilities, and local environmental setting data and parameters for the seven major generator/storage sites considered in this analysis. Additional site-specific environmental setting information is presented in Appendix C.

3.2.1 Hanford Site

Hanford occupies an area of about 1,450 km² of land desert in southeastern Washington State, about 191 km southwest of the city of Spokane and 240 km southeast of the city of Seattle (Figure 3.2). Located in Benton, Franklin, and Grant counties, Hanford is about 50 km long north to south and 40 km wide east to west. It is located north of the confluence of the Snake and Yakima rivers with the Columbia River. The Columbia River flows through the northern part of Hanford and turns south, forming part of Hanford's eastern boundary. The Yakima River, along part of the southern boundary, joins the Columbia River below the city of Richland, which bounds Hanford on the southeast. Rattle-snake Mountain, Yakima Ridge, and Umtanum Ridge form the western and southwestern boundaries of Hanford. The Saddle Mountains are located on Hanford's northern boundary. The cities of Richland,
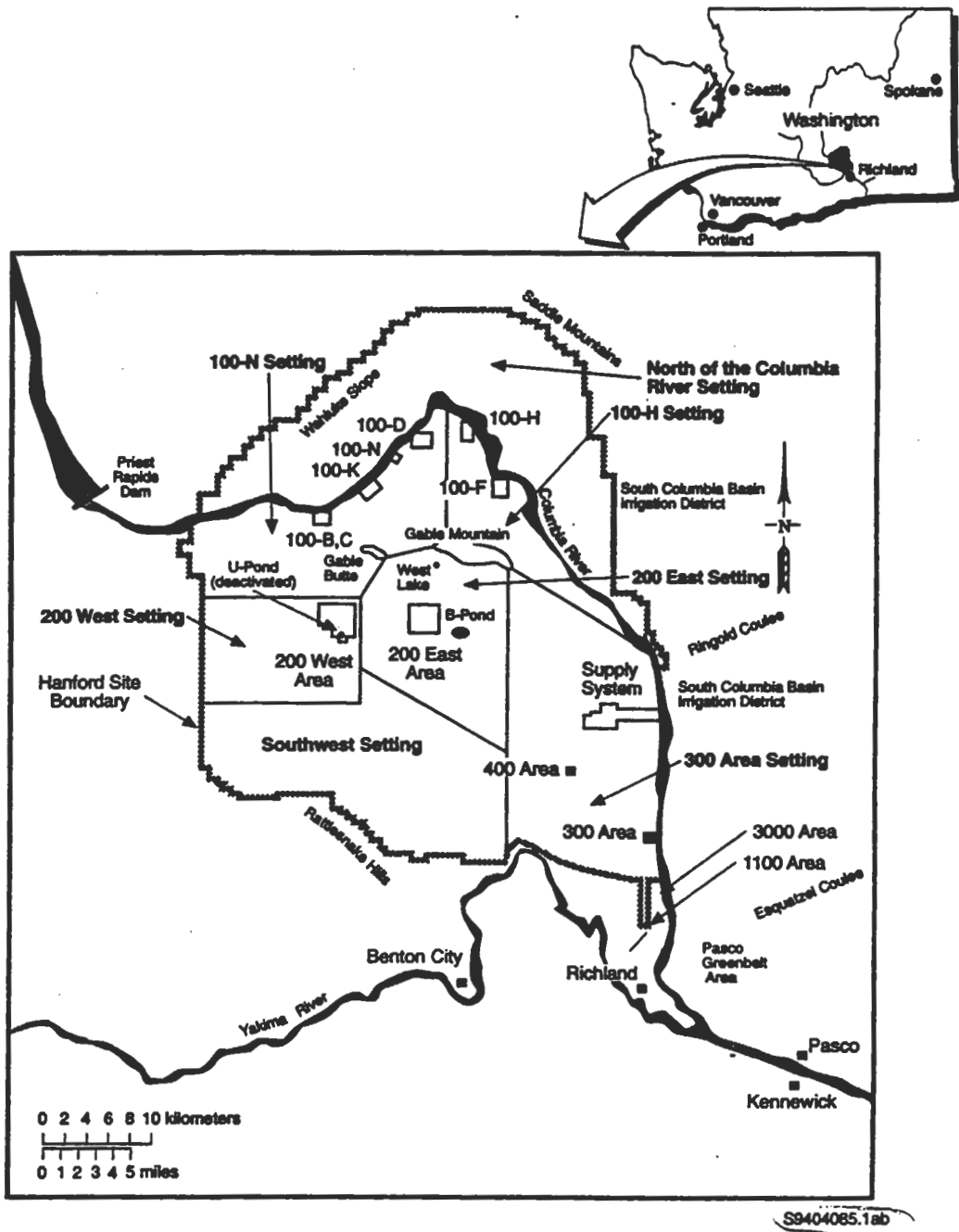
Figure 3.2. Location and Environmental Settings for the Hanford Site

Kennewick, and Pasco (commonly referred to as the Tri-Cities) are the nearest population centers (Pacific Northwest Laboratory 1992). The northern border of the city of Richland extends to Hanford's southern border near the 300 Area.

The site is located on the intermontane Columbia Plateau, with topography dominated by basalt plateaus, ridges, and buttes. Major rock units are from oldest to youngest: the Columbia River flood basalts and the Ringold Formation, consisting of unconsolidated fluvial sediments, coarse-grained glacial sediments (referred to as the Hanford formation and also informally referred to as the Pasco gravels).

The uppermost unconfined aquifer at Hanford occurs primarily within sediments of the Hanford and Ringold Formations. The hydraulic conductivity in the Ringold Formation is more than ten times lower than in the Hanford formation (Thorne and Chamness 1992). Before wastewater disposal began at Hanford, the unconfined aquifer was almost entirely within the less-transmissive Ringold Formation sediments (Thorne and Newcomer 1992). However, wastewater-disposal operations in the 200-East and 200-West areas have created groundwater mounds at these locations, and the water table is now in the more-transmissive Hanford formation over much of the eastern portion of the site (Thorne and Newcomer 1992). As a result of the shutdown of Hanford production facilities, the water table is declining. This decline in the water table should continue until new equilibrium conditions are reached (Newcomer et al. 1992). Deeper confined aquifers are found within the Columbia River basalts.

The Columbia River is the only significant surfacewater at Hanford and forms a major discharge point for groundwater at the site. The Hanford Reach is the portion of the river that extends from Priest Rapids Dam (upstream of Hanford) to the head of Lake Wallula (downstream of Hanford). Another surfacewater feature in the vicinity of Hanford is the Yakima River. This river approaches Hanford's southwestern boundary and discharges into the Columbia River about 12 km south.

The TRU waste generated since 1970 have been retrievably stored within the two operational areas at Hanford (200-East and 200-West). The CH-TRU waste is generally stored on asphalt pads in steel drums separated by plywood and plastic-reinforced nylon sheeting and are covered with about 1.2 m (4 ft) of soil to reduce direct radiation exposure. The RH-TRU waste is stored either in caissons or packaged with sufficient shielding to meet the external dose rate requirements for CH-TRU. Waste unsuitable for asphalt pad or caisson storage because of size, chemical composition, security requirements, or surface radiation have been packaged in reinforced wood, concrete, or metal boxes and stored in dry waste trenches.

3.2.2 Idaho National Engineering and Environmental Laboratory

The INEEL site is located in the southeastern part of Idaho, approximately equidistant from Yellowstone National Park; Salt Lake City, Utah; and Boise, Idaho (Figure 3.3). The INEEL is situated along the western edge of the eastern Snake River Plains at an elevation of 1,500 m and
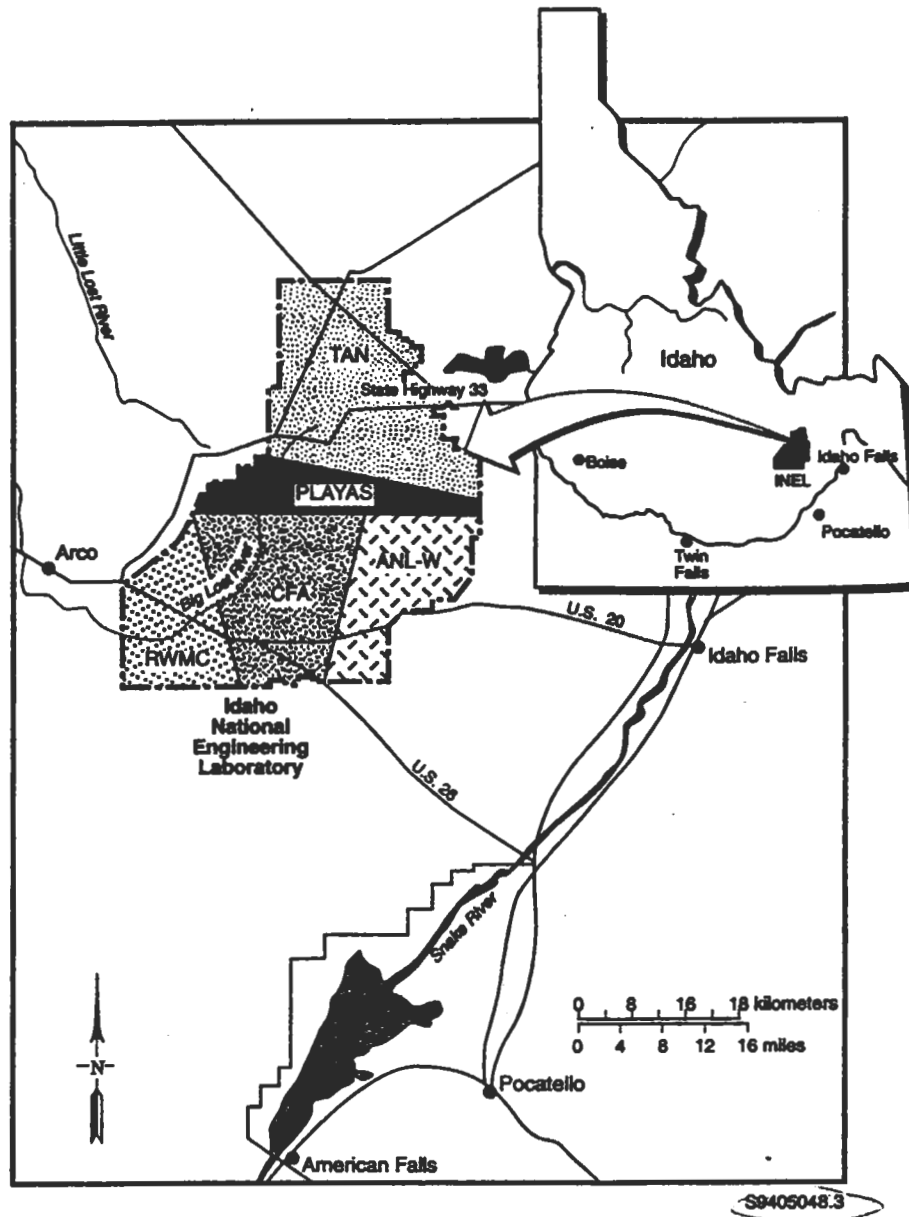
Figure 3.3. Location and Environmental Settings for the Idaho National Engineering and Environmental Laboratory

encompasses a desert area of about 2,300 km². The site extends 63 km from north to south and is about 58 km wide at this broader southern part. The nearest major community is Idaho Falls (population about 46,000), located about 64 km east of INEEL.

The TRU waste at INEEL has been stored within the Radioactive Waste Management Complex (RWMC). More complete descriptions of the INEEL, the RWMC, the TRU waste-storage and -examination facility, and the TRU waste stored on the Transuranic Storage Area pads can be found in DOE (1986). TRU waste generated since 1970 has been segregated from other wastes within the

3.11

complex and are stored on asphalt pads in steel drums, steel boxes, and fiberglass-covered wooden boxes. Some waste buried at the RWMC before 1970 is contaminated with TRU radionuclides. For this analysis TRU waste was assumed to be stored in the RWMC setting (see Figure 3.3).

3.2.3 Lawrence Livermore National Laboratory

The LLNL occupies 3.3 km^2 in the Livermore Valley, a depression within the Diablo Range of the California Coast Range Province (Figure 3.4). LLNL has two separate sites: the main Livermore site and Site 300. The main site is located near the eastern boundary of the city of Livermore, California, about 55 km east of downtown San Francisco. Site 300 is located in the Altamont Hills of the Diablo Range, about 24 km east of the main site. The area surrounding LLNL is used for residential, industrial, commercial, and agricultural activities.

Small amounts of mixed wastes (about 214 m^3) currently are stored at the LLNL main site as a result of laboratory operations in connection with nuclear weapons research and development. A small percentage of these wastes are mixed RH-TRU waste that are subject to land-disposal restriction requirements. The mixed TRU waste consists of 1) inorganic debris, which includes scrap metal, metal shavings, and lead bricks; 2) aqueous liquids from metal etching; and 3) organic liquids, such as spent solvents. For this analysis, TRU waste was assumed to be stored in the main setting (see Figure 3.4).

3.2.4 Los Alamos National Laboratory

The LANL empasses 110 km^2 in northcentral New Mexico and is located on long, narrow mesas, whose tops range in elevation from 2,400 on the flanks of the Jemez Mountains to about 1,900 m at their eastern termination above the Rio Grande Valley. LANL is situated about 97 km north-northeast of Albuquerque, New Mexico, and 40 km northwest of Santa Fe (Figure 3.5).

Since 1971, TRU waste has been packaged and stored in either subsurface trenches or aboveground earthen berms at the waste-burial site. Two types of packaging generally have been used. Small items have been stored in 208-L steel drums (sealed and coated with bituminous corrosion-protection material), and larger items have been placed in plywood crates (sealed and coated with fiberglass-reinforced polyester). Plywood storage crate sizes vary considerably, with a maximum length of about 9 m. For this analysis TRU waste was assumed to be stored in the Mesa Tops setting (see Appendix C).

3.2.5 Oak Ridge National Laboratory

The ORNL is located on 36 km^2 in the southcentral portion of the 140-km^2 Oak Ridge Reservation (ORR) (Figure 3.6). The ORR is located about 32 km west of Knoxville, Tennessee, in the rolling terrain between the Cumberland and Great Smoky mountains. The land surrounding the ORR is primarily rural, dominated by agricultural and residential land. The main laboratory area (designated as the X-10 facility) takes up only about 15% of the ORR grounds.
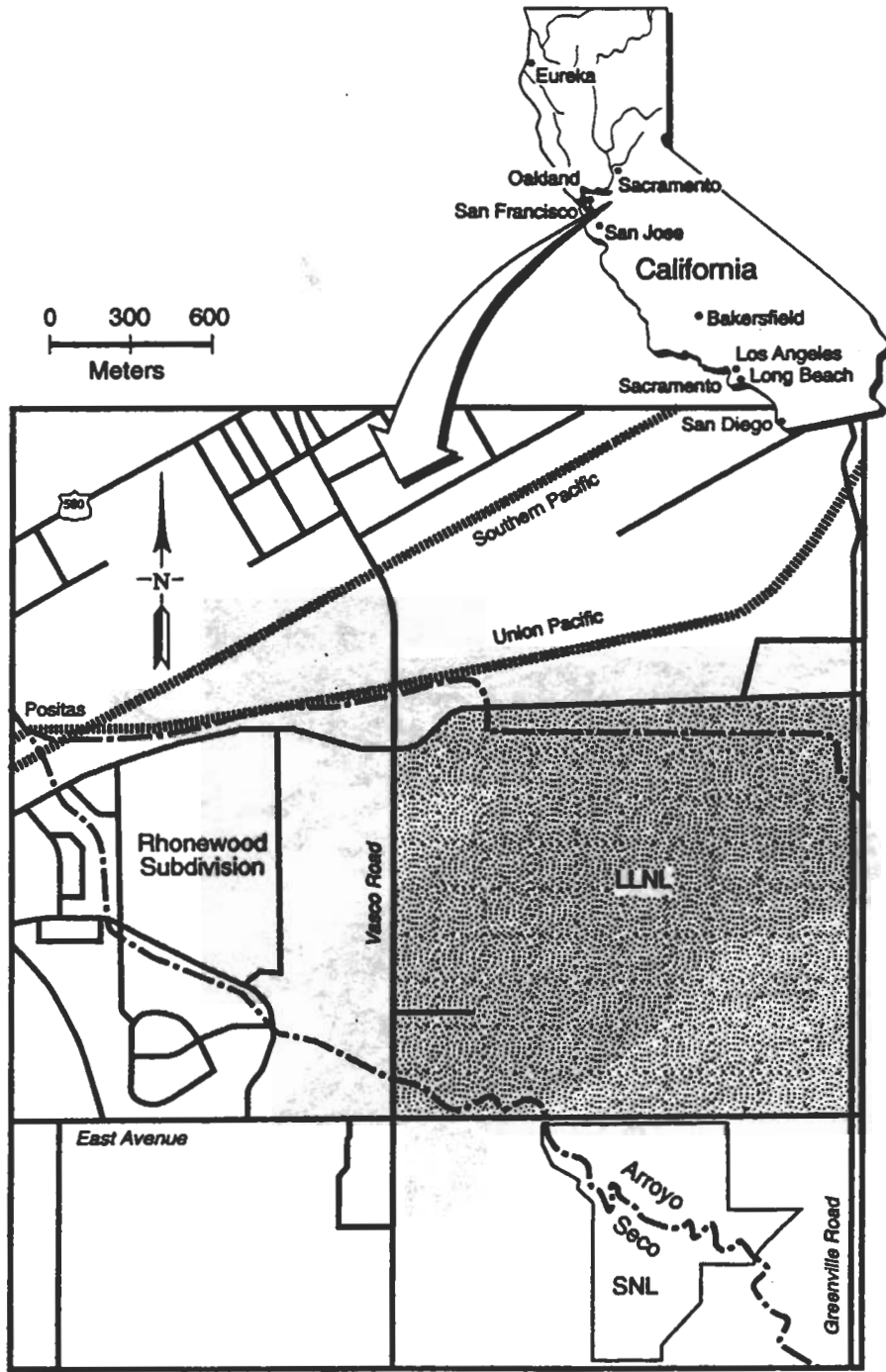
Figure 3.4. Location and Environmental Settings for the Lawrence Livermore National Laboratory

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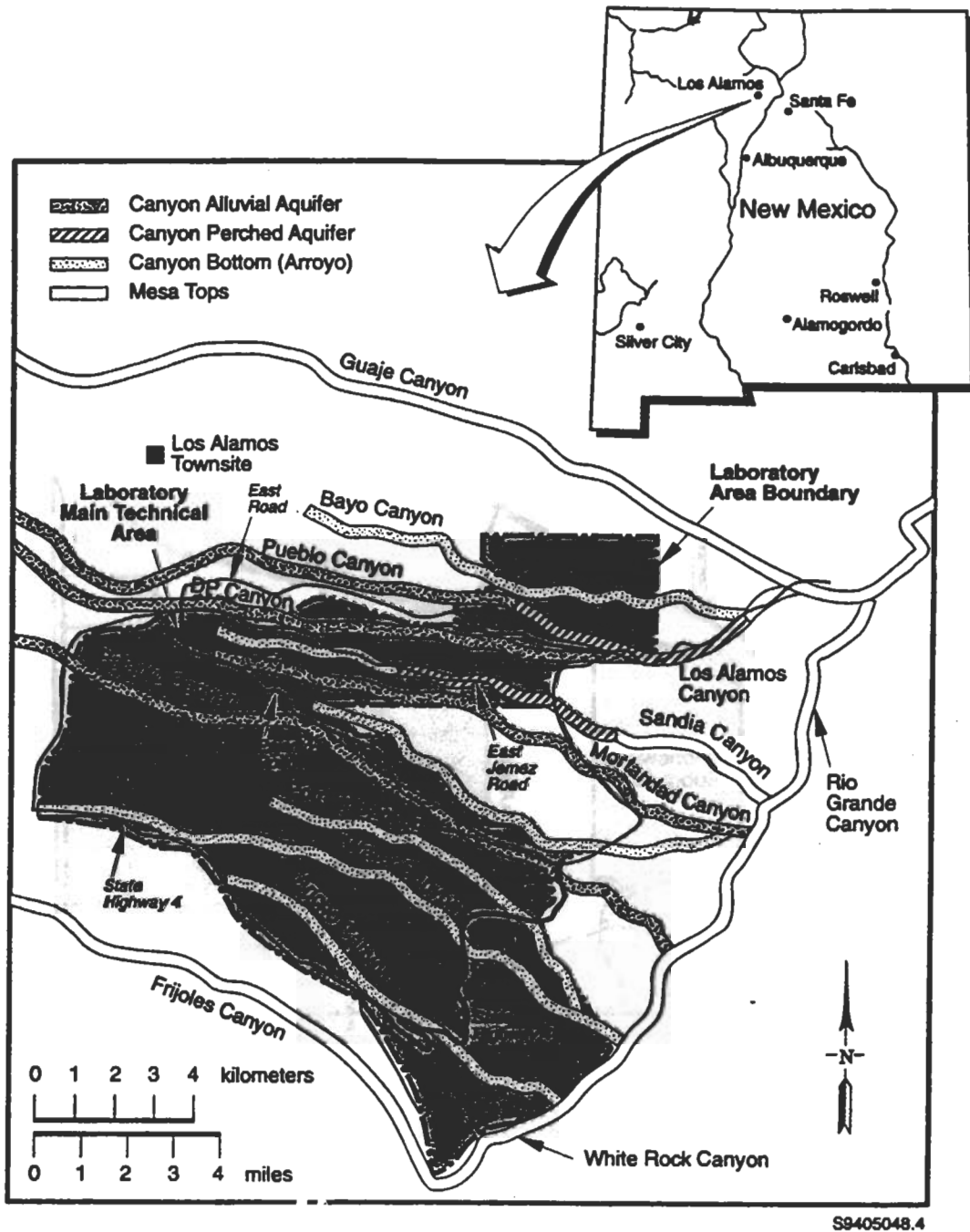
Figure 3.5. Location and Environmental Settings for Los Alamos National Laboratory

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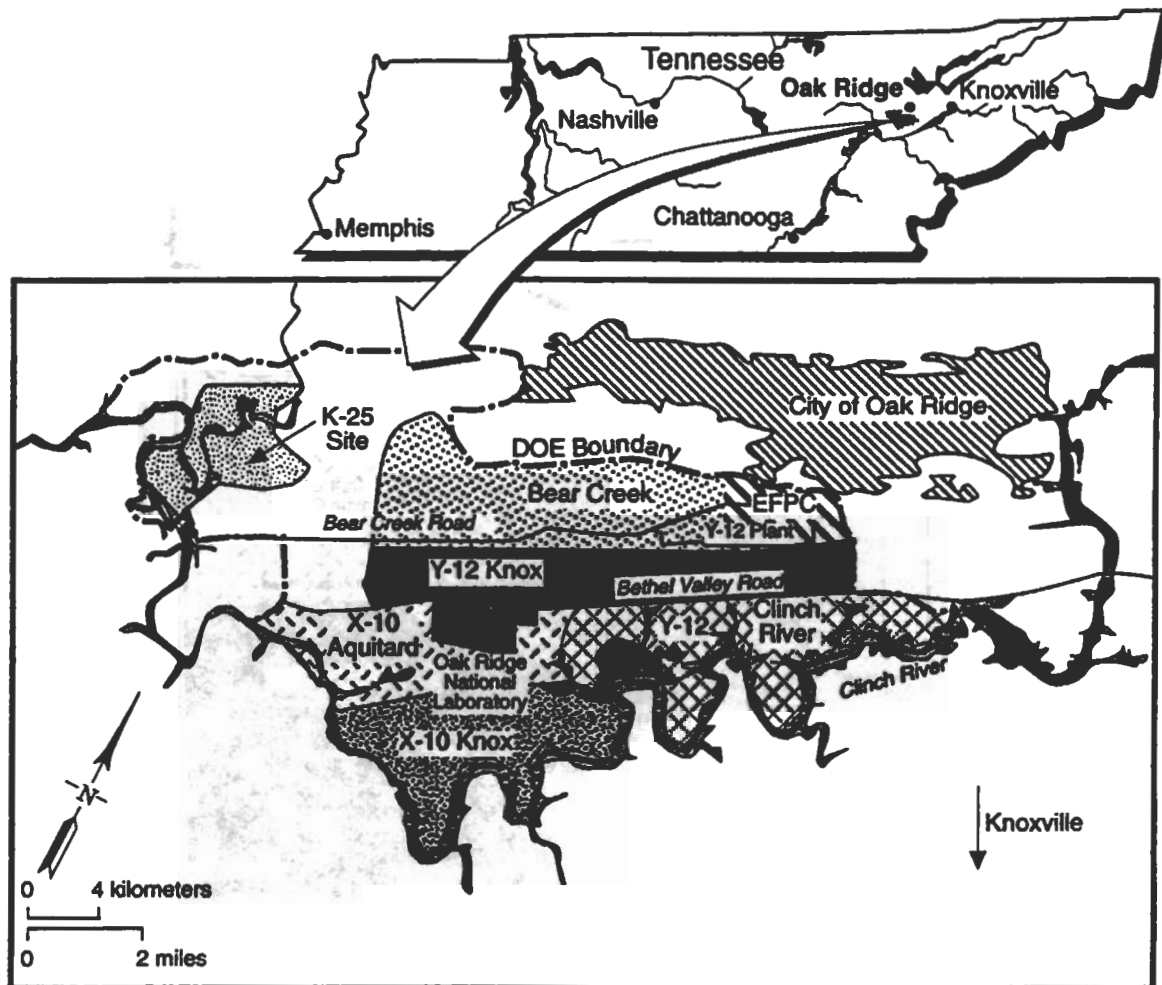
Figure 3.6. Location and Environmental Settings for the Oak Ridge Reservation

The TRU waste is generated in the main ORNL complex, primarily in the Isotopes Area and the Radiochemical Engineering Development Center. Newly generated CH-TRU waste is packaged in stainless steel drums at the point of generation and is transported within the site boundary to the TRU waste-storage area. For purposes of this analysis, TRU waste was assumed to be stored in the Knox environmental setting (see Appendix C).

3.2.6 Rocky Flats Environmental Technology Site

The RFETS is located east of the foothills of the Rocky Mountains about 26 km northwest of Denver, Colorado (Figure 3.7). The RFETS is situated in a generally rural area outside the Denver metropolitan area, with some ranches and industrial facilities located nearby. The installation occupies 27 km^2, with its facilities centrally located on only about 6% of the property. The topography of the RFETS is relatively flat.

3.15
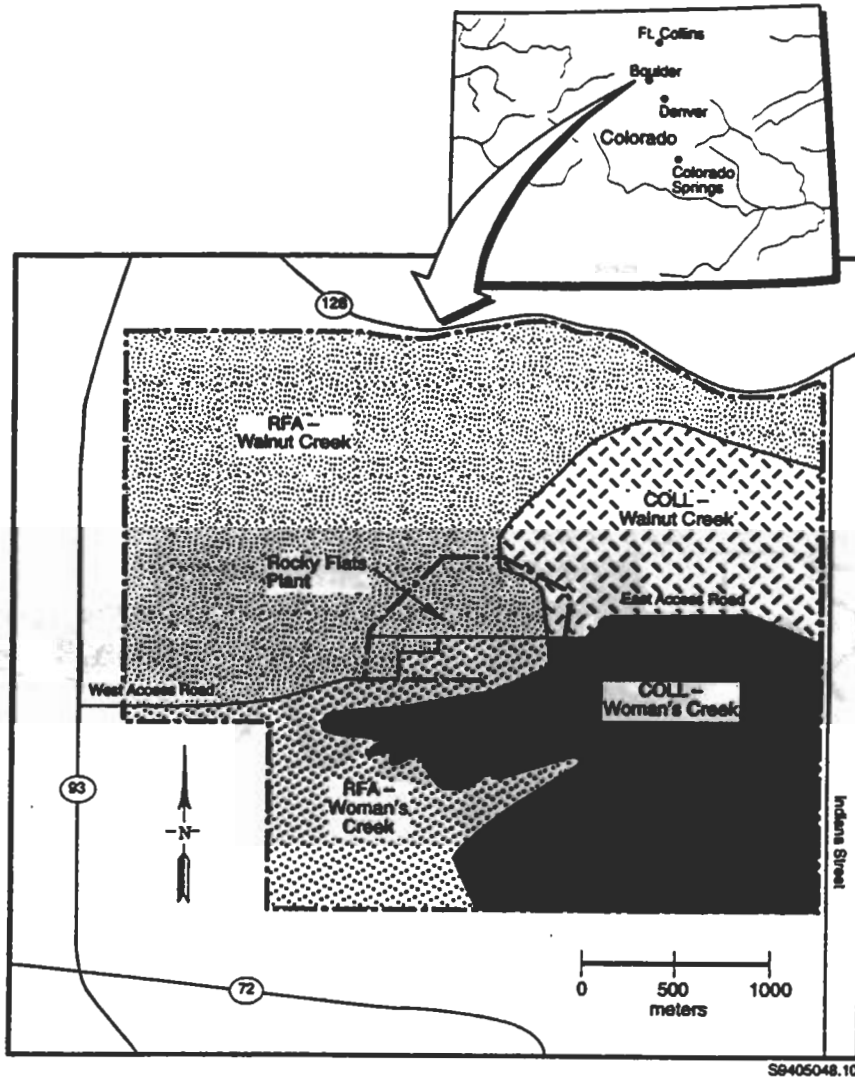
Figure 3.7. Location and Environmental Settings for the Rocky Flats Environmental Technology Site

The RFETS has two major surface drainages. The northern half of the installation drains into Walnut Creek, a low-flow stream that flows through a series of small ponds on the plant property before eventually discharging into the Great Western Reservoir several kilometers east of the RFETS. Flow in the creek is primarily storm flow. Natural drainage on the southern half of the property feeds into Woman's Creek. Much of the storm flow and runoff from the facilities area is diverted to a drainage ditch that parallels Woman's Creek. Waters from the southern half of the property discharge to Standley Lake, a major surfacewater body located south and east of the installation.

The Rocky Flats Plant Supercompaction and Repackaging Facility and TRU Waste Shredder would process solid waste that is newly generated during routine production operations, maintenance activities, and laboratory-support operations and may process waste in permitted storage. The Colorado Department of Health currently recognizes eight permitted storage areas at the RFETS for TRU mixed

waste. The areas differ in size for a total permitted storage capacity of 1,224 m³. The storage units are within existing structures having concrete floors covered with epoxy paint and fenced areas within the buildings, which allow segregation of the storage facility from adjacent operations. For this analysis TRU waste was assumed to be stored in the Operable Unit 3 setting.

3.2.7 Savannah River Site

The SRS is located in the southwestern portion of South Carolina, about 20 km south of Aiken, South Carolina, and 40 km southeast of Augusta, Georgia (Figure 3.8). The installation occupies 800 km², with about 10% of the property being used to support facilities and operations (Arnett et al. 1993).

The SRS is bounded on the south by the Savannah River for about 32 km, about 260 km upriver from the Atlantic Ocean. Five major streams at the SRS feed into the Savannah River: Upper Three Runs Creek, Four Mile Creek, Pen Branch, Steel Creek, and Lower Three Runs Creek. Other sur- facewater bodies at the site that feed into the five streams include PAR Pond, L Lake, Tims Branch, Tinker Creek, and Meyers Branch (Arnett et al. 1993). In all but one environmental setting developed for the SRS (the P-R Area setting), surface and subsurface drainage from both upland and lowland areas flow into creeks flowing southward into the Savannah River.

The TRU waste at the SRS is in retrievable storage on concrete pads or buried in shallow trenches. The waste is contained in concrete and steel boxes, concrete culverts, and galvanized steel drums covered with 1.2 m of soil or tornado netting (in use since 1985). For this analysis, TRU waste was assumed to be stored in the GS-TNX-D Area setting (see Appendix C).

3.3 Contaminant Source Term

This section describes the inventory of radionuclides and hazardous chemicals used in the long-term, lifetime exposure analysis. The initial inventory considered was the SEIS-II No Action Alternative 2 Basic Inventory, described in Section 3.0 and Appendix A of the WIPP SEIS-II. However, the radionuclide inventories have been decayed an additional 100 years from those shown in the WIPP SEIS-II to account for the radiological decay occurring during the institutional control period.

Inventory information from Appendix A of the WIPP SEIS-II showed there were 61 radionuclides, four inorganic substances, and 10 organic compounds that could be present in TRU waste at the various generator/storage sites. To concentrate data-collection efforts and analysis time on those contaminants that would contribute most to associated site hazards, screening analyses were conducted for the different contaminant groups. Sections 3.3.1, 3.3.2, and 3.3.3 describe the screening that was performed for radionuclide, heavy metal, and organic compound inventories, respectively.
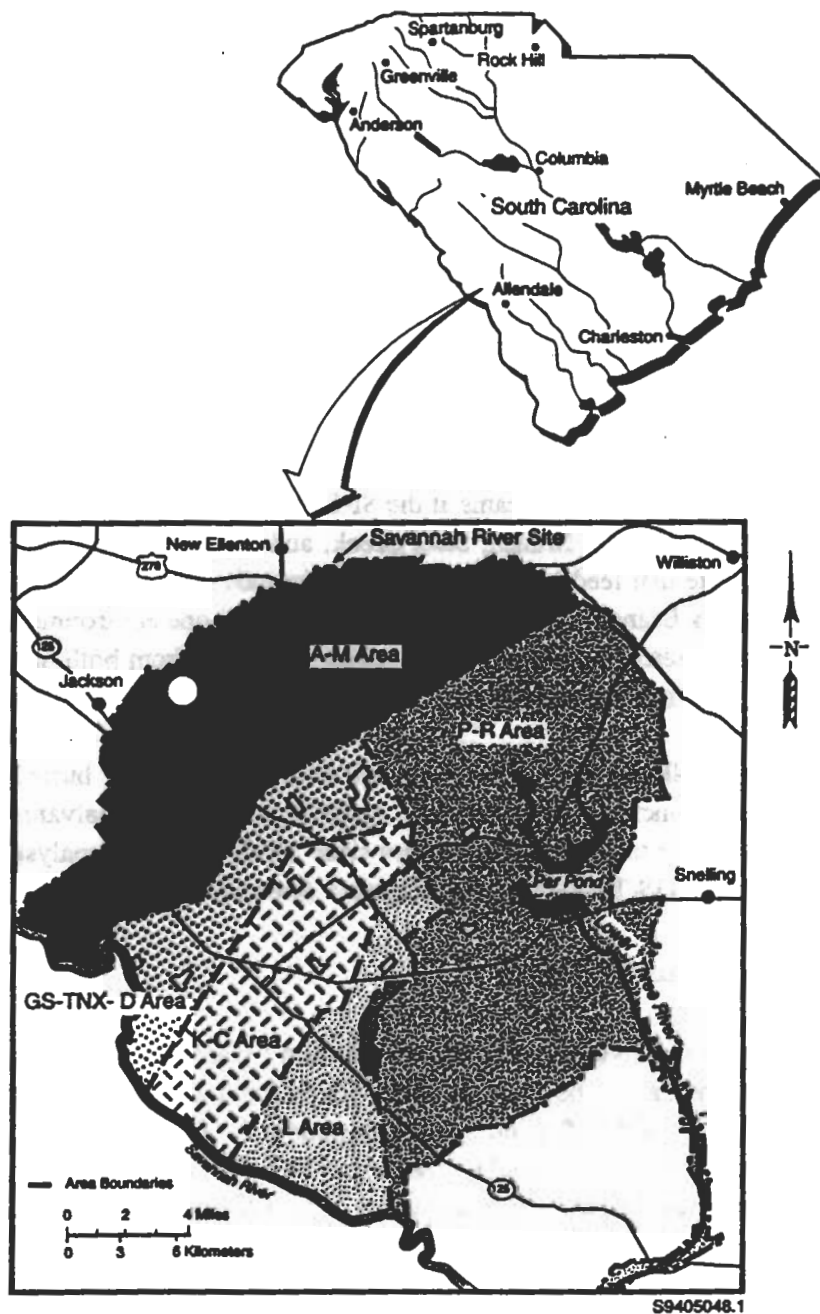
Figure 3.8. Location and Environmental Settings for the Savannah River Site

The complete list of contaminants evaluated in long-term, lifetime exposure analyses is shown in Table 3.2. The screening analyses resulted in a total of 32 contaminants of concern: 18 radionuclides, four heavy metals, and 10 organic compounds. Radionuclides were further screened for importance in the air and water pathways. Once this list was developed, an effort was made to obtain improved values for certain contaminant properties at specific sites. The contaminant-diffusion coefficient in

3.18

Table 3.2. Contaminants Included in Long-Term Lifetime Exposure Analyses

Contaminant of Concern	Type			
	Airborne Radionuclide	Waterborne Radionuclide	Heavy Metal	Organic
Americium-241	x			
Americium-243	x			
Carbon-14		x		
Curium-243		x		
Curium-244	x	x		
Cesium-137	x			
Europium-152	x			
Neptunium-237		x		
Protactinium-233	x			
Plutonium-238	x	x		
Plutonium-239	x	x		
Plutonium-240	x	x		
Plutonium-241	x			
Radium-226	x			
Strontium-90	x			
Uranium-233		x		
Uranium-234		x		
Uranium-235		x		
Lead			x	
Beryllium			x	
Cadmium			x	
Mercury			x	.
Carbon tetrachloride				x
Chloroform				x
Methylene chloride				x
1,1 Dichloroethylene				x
Methyl ethyl ketone				x
1,1,2,2-Tetrachloroethane				x
Toluene				x
Chlorobenzene				x
1,2 Dichloroethane				x
1,1,1-Trichloroethane				x

porous cement, required to model contaminant release from cemented TRU waste forms, and contaminant solubility, required to model the solubility bounding case, were evaluated (see Appendix B). Once the updated set of contaminant parameters was developed, actual waste-zone contaminant-release calculations were performed, and flux factors were generated.

3.3.1 Radionuclide Screening

The No Action Alternative 2 Basic Inventory radionuclide inventory (decayed an additional 100 years) for CH- and RH-TRU waste is presented in Tables A-36 and A-37 in WIPP SEIS-II Appendix A. The screening analysis for radionuclides examined separately transport and exposure by the air and water pathways. For the air pathway analysis, it was assumed that unit amounts of waste, soil, or debris, with radionuclide concentrations proportional to their inventories, were suspended by wind and transported through the air to a hypothetical receptor. The radiation dose resulting is a function of the radionuclide inventory and radionuclide-specific inhalation dose factor. Radionuclides were ranked by their contribution to the radiation dose. The largest individual radionuclide contributors and all those contributing 90% of the total screening dose were designated as the airborne radioactive contaminants of concern. These radionuclides are shown in Table 3.2.

Leaching through the vadose zone to the groundwater was the second screening pathway considered. Contaminants must be present in large enough quantities to potentially affect human health through this pathway. Also, the site must have the necessary climatology and surface-soil characteristics to percolate enough water to leach the contaminant from the waste zone and transport it through the vadose zone. The contaminant then must be transported through the aquifer to a groundwater well, which feeds the associated surfacewater medium. There, it must be present in high enough concentrations, with sufficient toxicity, to impact human health. Finally, radioactive contaminants must have long enough half-lives to sustain significant quantities of the contaminant during the time required for transport.

To conduct this screening, slightly simplified MEPAS runs were made. These runs used all the release-site data and assumed unit inventories for each contaminant. The release from the waste zone was assumed to be controlled by contaminant K_d values (i.e., the release was from a soil/debris TRU waste form and was not solubility limited). Transport through the environment also was controlled by the contaminant K_d values. Because of the importance of the assumed K_d values, all MEPAS-generated radionuclide K_d values were reviewed and improved with site-specific data, where available (see Appendix B, Table B.3). These MEPAS runs produced unit risk factors for each contaminant/site pair that were multiplied by the site-specific inventory to establish the estimated risk for each contaminant at each site. Relative risks for each radionuclide were again compared and ranked. Radionuclides that contributed 99% of the total relative risk were designated as waterborne contaminants of concern.

3.3.2 Heavy Metal Screening

The No Action Alternative 2 Basic Inventory heavy metal inventory is shown in Table A.5. Reliable inventory data generally were not available for the heavy metals. As a result, a qualitative screening method was used.

Lead shielding is used to reduce surface-dose rates to acceptable levels for RH-TRU waste containers. Lead, therefore, becomes a major part of the total waste mass and is included on the list of contaminants of concern. Lead from personal protective equipment also is a major contaminant in CH-TRU waste. Inventory estimates for beryllium, cadmium, and mercury were available from DOE (1995a). These contaminants were added to the list of contaminants of concern as well. Other heavy metals, such as chromium, were not included on the list because of the lack of available inventory data.

3.3.3 Organic Contaminant Screening

The No Action Alternative 2 Basic CH- and RH- TRU Inventory for organic contaminant is shown in Tables A.6 and A.7. As with inorganic contaminants, little reliable inventory data were available for the organic contaminants. Some data reported in DOE (1996a), however, could be used as an indirect indication of volatile organic concentrations. DOE (1996a) summarizes the results of a head-space-sampling and -analysis study conducted on TRU waste from INEEL and RFETS. That study sampled approximately 930 drums of varying waste types to determine a weighted-average head-space/gas composition that could be used for all TRU waste. The weighted values were screened using the concentration toxicity-screening technique presented by the U.S. Environmental Protection Agency (EPA) (1989). This screening provided a list of carcinogenic and noncarcinogenic contaminants that accounts for over 99% of the human-health risk resulting from migration in air.

This list of contaminants of concern was adopted as the organic contaminants of concern, with one exception; carbon disulfide was not actually detected in any of the head-space analyses. Carbon disulfide is presented in this analysis in cases where no data were available because one-half of the calculated sample quantitation limit was used as the assumed head-space concentration. Even so, carbon disulfide accounted for <2% of the noncarcinogen human-health impact. As a result, carbon disulfide was not included as an organic contaminants of concern. Data were available for two other contaminants: 1,2-dichloroethane and 1,1,1-trichloroethane. These were added to the eight screened contaminants to give a total of 10 organic contaminants of concern.

3.3.4 Additional Inventory for Radionuclides

In addition to the No Action Alternative 2 Basic Inventory, the potential impacts of radionuclides in the SEIS-II Additional Inventory were evaluated for contribution to the aggregate impacts from all sites over 10,000 years. The Additional Inventory was not analyzed in the SEIS-II No Action Alternative 2 but was analyzed for Action Alternatives 1, 2, and 3. This inventory includes TRU waste, mainly buried waste, that currently exists and would remain at the sites under No Action Alternative 2. Under

No Action Alternative 2 it would not be treated or undergo any additional management. Table A.8 shows the CH- and RH-TRU Additional Inventory for radionuclides by site, decayed an additional 100 years to account for the institutional control period.

3.4 Waste Configuration and Release Scenarios

This section describes the assumptions used in the source-term-release analysis, which provides the contaminant-flux factors in the MRA methodology approach. Topics include a general conceptual site-model for buried and surface-stored TRU waste, assumptions governing the release of contaminants from the TRU waste-form categories, contaminants of concern considered in this analysis, and sample outputs resulting from source-term-release calculations. The contaminant-flux factors developed for the buried and surface-stored TRU waste are used to evaluate human health impacts from atmospheric, groundwater, and surfacewater media associated with each of the seven generator/storage sites.

3.4.1 Conceptual Model for Contaminant-Release Scenarios

The overall geometric configuration of the waste-storage zone, the assumed degradation of the waste-storage zone, and how the distribution of the TRU waste forms within the zone affect the magnitude and areal extent of the contaminant-release fluxes from the zone are discussed below. These configurations and assumption are important in defining the source and the release of TRU contaminants into the environment. Based on these data, the contaminant-release fluxes can be defined for both surface and buried waste sites that are critical for defining transport and exposure scenarios.

3.4.2 Waste-Storage Configuration

The following assumptions were made for the analysis of buried TRU waste: 1) all TRU waste is contained in 55-gal (208-L) drums (91.4 cm tall) stored together in one shallow burial zone; 2) four layers of drums are stacked on an asphalt or concrete pad, with plywood sheets between the layers and on top; and 3) 1.2 m of contaminant-free soil are backfilled over the layers of drums. This overall configuration is illustrated in Figure 3.9.

For the analysis of surface-stored TRU waste, the initial waste-zone configuration is similar to that for buried TRU waste. It is assumed that four layers of drums are stacked on an asphalt or concrete pad, with plywood sheets between the layers and on top. The stacked drums are placed in metal storage buildings, instead of being buried. This overall configuration is illustrated in Figure 3.10.

The vertical dimension of the waste zone for both buried and surface-stored configurations is approximately 3.7 m. The drums are placed as closely to one another as is possible in a rectangular
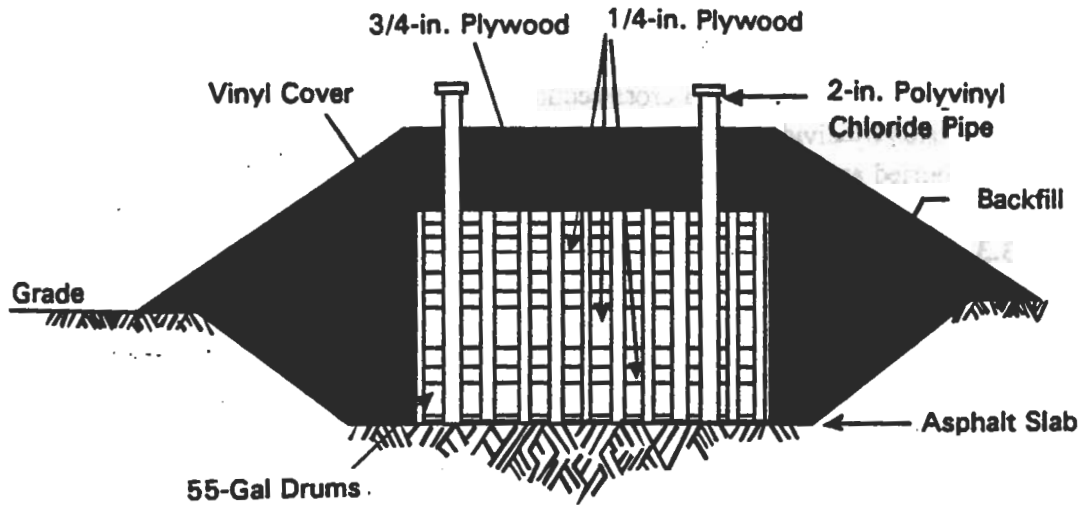
Figure 3.9. Overall Configuration of the Buried Waste-Storage Zone
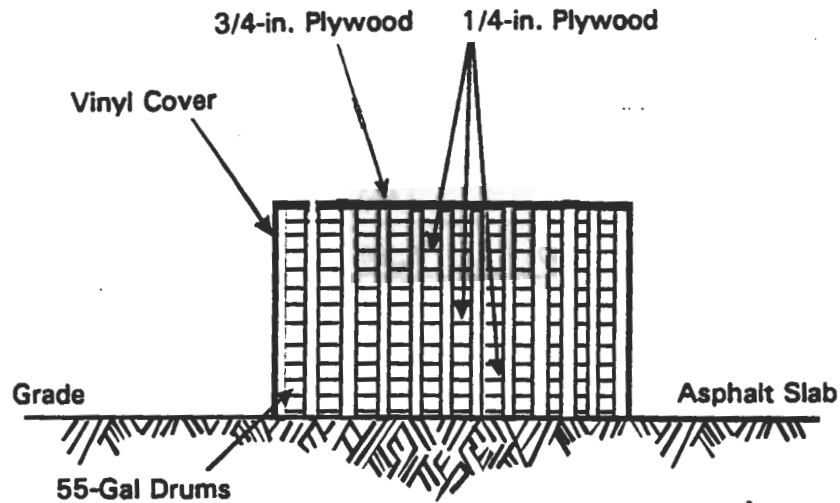
Figure 3.10. Overall Configuration of the Surface Waste-Storage Zone

grid arrangement. Because of this configuration, the volume of the waste zone is larger than the volume of the TRU waste itself. The ratio of waste-zone and waste-drum volumes for a rectangular grid arrangement is as follows:

$$[N(2r)^2(4h)/(N\pi r^2(4h)] = 4/\pi \tag{3.9}$$

where N = number of drums
       r = drum radius
       h = drum height.

Waste-zone volumes for each TRU waste-form category are calculated for each site by multiplying the corresponding volumes reported in Table A.5 by $4/\pi$. These overall waste-zone volumes are reported in Table 3.3. The horizontal cross-sectional areas of each TRU waste-form category for each site can be calculated by dividing the waste-zone volumes by 3.7 m. These areas, along with the relative amounts of buried and surface stored waste for each site, also are shown in Table 3.3.

Table 3.3. Overall Waste-Zone Volumes and Horizontal Areas for each Waste-Form Category by Release Site

	Volume (m^3)		Area (m^2)		Zone Configuration (%)	
Release Site	Soil/Debris	Cement	Soil/Debris	Cement	Buried	Surface
Hanford 200E	55,300	0	15,200	0	13.6	86.4
Hanford 200W	55,300	0	15,200	0	13.6	86.4
INEEL	33,400	6,230	9,150	1,710	49.9	50.1
LANL	17,300	9,800	4,740	2,690	37.9	62.1
LLNL	1,480	39.7	406	10.9	0.0	100.0
ORNL	5,350	1,680	1,470	460	15.2	84.8
RFETS	10,700	3,090	2,940	848	0.0	100.0
SRS	13,091	2,280	3,590	624	11.6	88.4

3.4.3 Assumed Degradation

The ability of storage buildings, waste-configuration components, waste containers, and TRU waste forms to contain contaminants needs to be considered when modeling the long-term release of contaminants. The following discussion provides an overview of these considerations in terms of their effect on the long-term release of contaminants from TRU waste.

3.4.3.1 Facility

The surface-storage scenario assumes that TRU waste is housed in metal storage buildings. These buildings would degrade relatively quickly over the 10,000-year evaluation period. This chiefly is due to the lack of maintenance after the loss of institutional control. Therefore, metal storage buildings and enclosures were assumed to offer no protection, and the surface-storage scenario was modeled as if the stacked waste drums were not sheltered for the 10,000-year period. Facility degradation was not applicable to the buried TRU waste scenario.

3.4.3.2 Waste Configuration

Components of the TRU waste-form configuration were assumed to degrade quickly relative to the 10,000-year evaluation period. Degradation of the plywood or the storage building would allow the spaces between the drums to fill with soil from the surface layer of the site. Likewise, once storage buildings degrade, the drums themselves would degrade at an accelerated rate, further altering the waste configuration.

It also was assumed that the asphalt or concrete pad would be cracked or otherwise degraded for essentially the entire 10,000-year evaluation period. This would allow infiltrating water to percolate through the waste zone, pass through the cracked or degraded pad, and move through the remainder of the vadose zone directly beneath the waste zone.

Volatilization losses of organic contaminants were assumed to be zero in the buried-waste scenario to establish a bounding case for groundwater contamination. In contrast, holes readily would develop in waste drums in the surface-stored scenario, causing most of the volatile organic inventory to be lost through volatilization. Therefore, the entire organic inventory was assumed to release through volatilization during the first year in the surface-stored scenario.

3.4.3.3 Waste Containers

The integrity and longevity of the drums also would be a factor in contaminant release from the waste zone. Both the surface-stored and buried scenarios assumed that TRU waste would be contained in mild steel, U.S. Department of Transportation 17-c drums. Corrosion rates for mild steel drums are quite high, even when buried in favorable, dry environments. Drum lives of less than 100 years would be expected. Once the storage enclosure or building degrades to the point where waste drums would be directly exposed to the elements, stored TRU waste drums would be expected to degrade more rapidly than buried TRU waste drums. Because the expected life of the waste drums is relatively short compared to the 10,000-year evaluation period, no credit was taken for containers in the analysis.

3.4.3.4 Cemented TRU Waste Form

The cemented TRU waste form was assumed to initially be a solid block having the same size and shape as a 55-gal (208-L) drum. Cemented monoliths are known to crack and degrade into porous material over time. Unfortunately, the theory for modeling the transition from a solid block to porous material and its effect on contaminant release is not well developed. It was assumed, therefore, that the cement waste blocks would remain intact for the first 500 years and then catastrophically fail. After failure, the waste zone was assumed to act as a porous material.

3.4.4 Distribution of TRU Waste Forms

The horizontal cross-sectional area of the source zone is a required modeling input parameter. In reality, all drums of a given TRU waste-form category would not be emplaced in a single location

within the waste zone. Rather, they would likely be interspersed with drums containing other TRU waste-form categories. To simplify the analysis, however, it was assumed that any specific areal location contains drums of only one TRU waste-form category over the four vertical layers. It also was assumed that the waste zone was composed of a random distribution of "reasonably large" subareas of drums of only one TRU waste-form category. Each subarea is of sufficient size so that contaminant release is controlled by the physics and chemistry of that subarea's TRU waste-form category alone. Therefore, contaminant release from the waste zone can be modeled in two parts (one for each waste-form category) using the conceptual mathematical models described in Section 3.1 and the appropriate fractional inventories and areas for each TRU waste-form category.

Contaminant mass flux is the output of source-term calculations for the subarea of each TRU waste-form category. Because the subareas were assumed to be uniformly dispersed throughout the waste zone, the mass flux of any contaminant from the two waste-form categories could be summed to determine the total mass flux of that contaminant over the cross-sectional area. Mass fluxes over the total waste-zone area are required inputs for subsequent transport simulations.

3.4.5 Contaminant-Release Scenarios

An overview of contaminant-release scenarios from the different TRU waste forms for both the surface-stored and buried waste configurations follows. Geochemical controls that may limit the contaminant release from the overall waste zone also are discussed.

The overall rate of contaminant-mass loss from the waste zone is the sum of five different processes: decay, leaching, wind erosion, water erosion, and volatilization. The buried-waste scenario assumes a 1.2-m (4-ft) cover layer of soil that significantly reduces TRU waste interaction with surface-erosion/dispersion mechanisms. Leaching and decay, therefore, are the only two loss processes considered for the buried-waste scenario.

A "multimedia" scenario was used for surface-stored releases. Water erosion, wind suspension, and volatilization were considered in addition to leaching and decay. This scenario assumes that there is no cover layer, and contaminant transport by water erosion and wind suspension begins at the start of the analysis. The effects of different TRU waste forms on each of the five different loss processes are discussed below.

3.4.5.1 Soil/Debris TRU Waste Form

The contaminants in soil/debris waste forms were assumed to be present either in the pore aqueous solution, in the pore vapor phase, or adsorbed to the porous medium particles. For radionuclides and metals, the contaminant also may be present in a solid phase (precipitates) if its inventory is higher than the capacity of the aqueous, vapor, and sorbed phases to contain it (i.e., solubility control). Similarly, for organic chemicals, the contaminant also may be present in an organic liquid phase (free product) if its inventory is higher than the capacity of the aqueous, vapor, and sorbed phases to contain it (i.e., solubility control). Unless this waste form is buried, all five loss processes potentially can occur.

Degradation/decay is assumed to be a first-order process. Therefore, the mass-flux equation for contaminant loss from the source zone by degradation/decay is given by:

$$\left[\frac{dM_i}{dt}\right]_{decay} = -\lambda_i M_i \qquad (3.10)$$

where $(dM_i/dt)_{decay}$ = contaminant-mass loss rate resulting from decay for contaminant I (g/yr or Ci/yr).

M_i = mass lost from decay for contaminant I (grams or curies).

t = time period of decay process (years).

λ_i = first-order degradation/decay coefficient for contaminant I (yr^{-1}).

When there is no solubility-controlled solid phase (for radionuclides and metals) or organic liquid phase (for organic chemicals) present, leaching loss of the contaminant is assumed to occur by desorption-controlled advective transport of the contaminant by previously clean water percolating through the porous waste. The waste zone is assumed to be a so-called "well-mixed reactor." Therefore, the mass-flux equation for contaminant release from the source zone by leaching in this instance is given by:

$$\left[\frac{dM_i}{dt}\right]_{leach} = -\frac{U\,M_i}{\theta_w R_{wi}(h_o - d_o)} \qquad (3.11)$$

where $(dM_i/dt)_{leach}$ = contaminant-mass loss rate resulting from leaching for contaminant I (g/yr or Ci/yr).

U = Darcy water-flux density of water infiltrating through the source zone (cm/yr)

θ_w = volumetric water content of the soil (dimensionless)

R_{wi} = standard retardation factor for contaminant I (dimensionless)

h_o = initial distance from soil surface to bottom of the contaminant-source zone (cm)

d_o = initial distance from soil surface to bottom of the clean soil layer (cm).

and retardation factor is defined by:

$$R_{wi} = 1 + \frac{\beta}{\theta_w} K_{di} \qquad (3.12)$$

where β = soil bulk density (g/cm^3)

K_{di} = linear equilibrium sorption coefficient for contaminant I (cm^3/g). The contaminant-specific linear equilibrium sorption coefficient used in this analysis is given in Appendix B (Table B.3).

when a clean layer of soil exists on top of the waste zone, and is given by:

$$\left[\frac{dM_i}{dt}\right]_{leach} = -\frac{U\,M_i}{\theta_w R_{wi}[h_o - (E + S)\,t]} \qquad (3.13)$$

where E = volumetric rate of soil removal by water erosion, per area (cm/yr)
       S = volumetric rate of soil removal by wind suspension, per area (cm/yr).

when the waste zone is exposed to the atmosphere, and water erosion and wind suspension are removing contaminated soil from the zone.

When the aqueous concentration of the contaminant is controlled by solubility, the mass-flux equation for contaminant loss from the source zone by leaching is given by:

$$\left[\frac{dM_i}{dt}\right]_{leach} = -\,U\,A\,C_{wi} \qquad (3.14)$$

where   A = horizontal cross-sectional area of the source zone (cm^2)
        C_{wi} = concentration of contaminant I in the aqueous phase (g/cm^3 or Ci/cm^3).

In this instance, C_{wi} is just equal to the solubility if the contaminant is a radionuclide or metal. However, if the contaminant is an organic chemical (present in an organic liquid mixture), C_{wi} is calculated by a more complicated algorithm that uses an analogue to Raoult's Law if the mole fraction of the contaminant in the liquid phase is close to one, an analogue to Henry's Law (i.e., the octanol-water partition-coefficient concept) if the mole fraction of the contaminant in the liquid phase is close to zero, and a linear interpolation for the region in between.

Water erosion and wind suspension are assumed to strip soil particles from the soil surface at a constant rate. If these processes are stripping away clean soil above a waste zone, their associated contaminant mass-flux terms are equal to zero. If the waste zone is exposed to the atmosphere, the mass-flux equations for contaminant loss from the source zone by water erosion and wind suspension are given by:

$$\left[\frac{dM_i}{dt}\right]_{erosion} = -\frac{E\,M_i}{h_o - (E + S)t} \qquad (3.15)$$

where $(dM_i/dt)_{erosion}$ = contaminant-mass loss rate resulting from water erosion for contaminant I
                                (g/yr or Ci/yr)

$$\left[\frac{dM_i}{dt} \right]_{suspension} = - \frac{S \, M_i}{h_o - (E + S) \, t} \qquad (3.16)$$

where $(dM_i/dt)_{suspension}$ = contaminant-mass loss rate resulting from wind suspension for contaminant I (g/yr or Ci/yr).

3.4.5.2 Cemented Waste Forms

For cemented waste forms, the distribution of contaminants between different phases within the porous cement is not accounted for explicitly in the theory. Decay and leaching are the only loss processes that are assumed to occur. Decay of the overall contaminant mass in the waste form is again assumed to occur according to Equation (3.10). Infiltration water percolating through the waste zone was assumed to not penetrate the waste forms. Rather, leaching loss is caused by this water, picking up contaminants as they diffuse through the water-filled pores of the cement from the interior to the waste-form surface. The mass-flux equation for contaminant release from the source zone by leaching (if diffusion from within the cement limits the release) is given by:

$$\left[\frac{dM_i}{dt} \right]_{leach} = - M_{io'} \, e^{-\lambda_i t} \left(\frac{A_c}{V_c} \right) \sqrt{\frac{D_{ci}}{\pi \, t}} \qquad (3.17)$$

where $M_{io'}$ = initial total mass or activity of contaminant I in the source zone for a cemented waste form (g or Ci)

A_c = total external surface area of the cemented waste forms in the contaminant source zone (cm^2)

V_c = total volume of the cemented waste forms in the contaminant source zone (cm^3)

D_{ci} = effective diffusion coefficient of contaminant I in a cemented waste form (cm^2/yr).

The phase distribution of the contaminant within the cement is implicitly expressed in the contaminant's effective diffusion coefficient in the cement. The contaminant-specific diffusion coefficients used in this analyses are given in Appendix B (Table B.4).

3.4.5.3 Geochemical Controls on Waste-Form Leaching

When the waste is a cemented waste form, leached contaminants do not immediately move out of the bottom of the waste zone. Because we assume that the space between these drum-shaped waste forms is filled with soil, the leached contaminants enter this soil zone before exiting the bottom of the waste zone with the percolating water. If the physical and chemical processes in this soil zone are such that contaminant leaching from the soil is slower than the leaching from the waste form itself, then this step of the release process is the limiting step. Therefore, the source-term-release module compares the leaching mass flux calculated by Equation (3.17) with the leaching mass flux calculated by

Equations (3.11) through (3.14), where it was assumed that the waste zone was composed of soil. If the release predicted by soil-desorption control or by solubility control is lower, this value is used for the leaching mass flux.

Source-term contaminant-release calculations were run for all 32 contaminants of concern for each site and for each waste-form category. If a contaminant was not present at a particular site, an inventory of zero was used for that calculation. Furthermore, if a particular TRU waste-form category was not present at a site, the relative fraction factor used for that TRU waste-form category was zero.

3.4.6 Example Contaminant-Release Calculations

This subsection presents sample contaminant-leaching flux versus time curves, illustrating the types of outputs from source-term-release calculations, differences in leaching-release behavior among the two TRU waste-form categories, and differences between soil-desorption and solubility controls. Actual analysis outputs were used as starting points for the example curves presented in Figures 3.11 and 3.12. Important features of the curves, however, were accentuated to better illustrate the general leaching behavior. Run parameters, such as contaminant inventory, half-life, solubility, and K_d, were adjusted. In addition, to enhance the effect of failure of the cemented TRU waste form, 50 rather than 500 years was assumed as the point at which the TRU waste-form failure occurs. Note also that the initial contaminant inventory and waste-zone size and shape are the same for all cases shown within a figure.
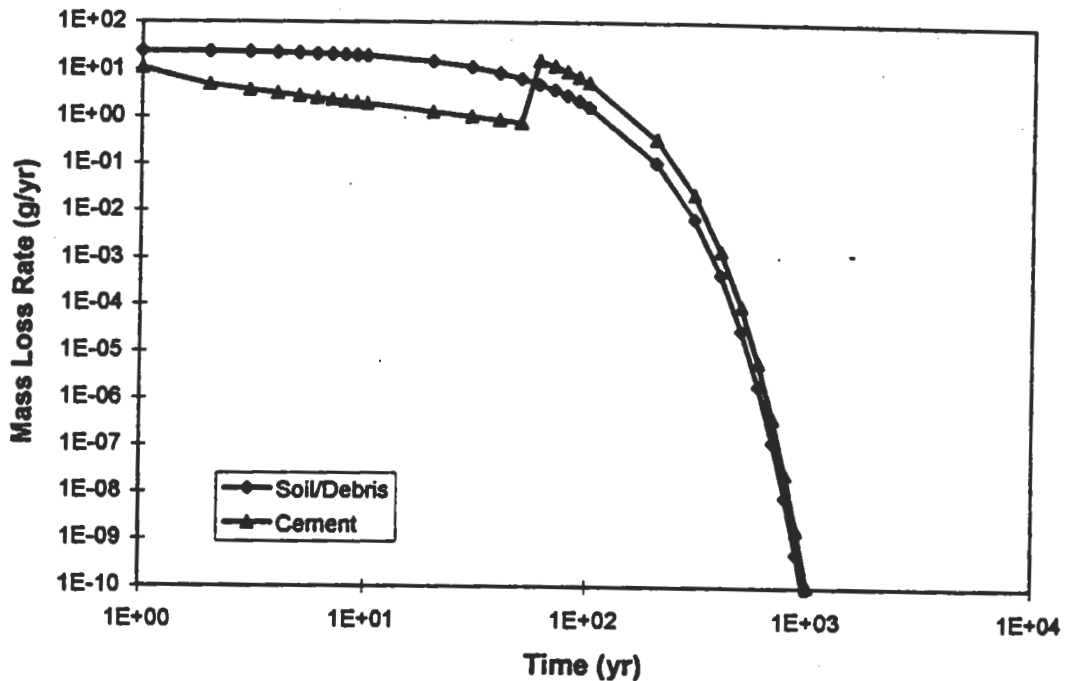
Figure 3.11. Example of Contaminant-Leaching-Flux Curves from Different Waste Forms

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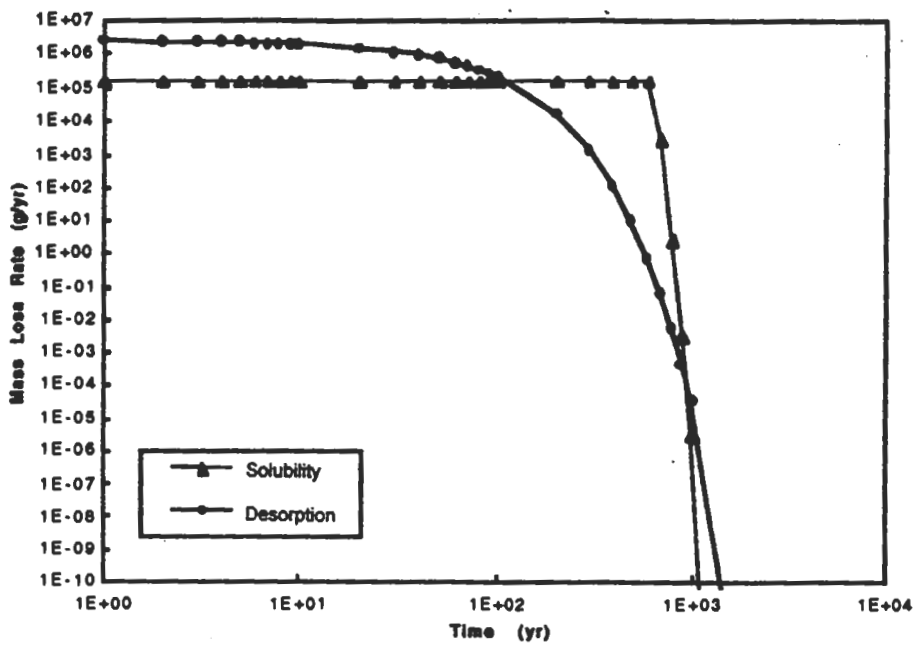
Figure 3.12. Example of Contaminant-Leaching-Flux Curves for Desorption and Solubility
Controlled Cores

Figure 3.12 is a logarithmic plot of contaminant-leaching flux versus time for the two TRU waste-
form types. Initially, contaminants are lost more rapidly from the soil/debris TRU waste form, and the
release curve for cement falls off more quickly. This is due to the diffusion-release mechanisms within
the cement waste form that set up contaminant concentration spatial gradients that inhibit the rate of
release over time. Eventually, however, the contaminant release from the soil/debris TRU waste form
also falls because the contaminant inventory in the soil/debris TRU waste form becomes exhausted.

Over time, release from the soil/debris TRU waste form continues to fall off monotonically,
whereas release from the cemented TRU waste form increases at the point when the waste form fails.
Because the initial contaminant release from cement is slower than that from soil/debris, more of the
contaminant remains in the cemented TRU waste form at the time of failure than remains in the soil/
debris TRU waste form at a corresponding time. The cemented TRU waste form is assumed to
degrade quickly into a porous material similar to the soil/debris waste form. At this time, the cemented
TRU waste form releases contaminants according to the same mechanisms as the soil/debris TRU waste
form. The higher contaminant inventory at the time of failure subsequently results in higher release
from the cement. Therefore, in the conceptual model adopted for this analysis, the cementing process
only delays contaminant release unless significant loss via radioactive decay occurs while the contami-
nant is held in the waste form.

3.31

Figure 3.12 compares contaminant-leaching flux curves for soil-desorption and solubility controls. Although the contaminant inventory used was different, the soil-desorption-control curve has the same features as the leaching curve for the soil/debris TRU waste form in Figure 3.11. For the desorption case, the solubility was set high enough, relative to the contaminant inventory, so that all the contaminant could reside in the aqueous and sorbed phases of the porous medium in the source zone.

The solubility value in the solubility-control curve was reduced such that excess contaminant would partition into a solid precipitate or organic liquid phase. In this case, the aqueous concentration of the contaminant will always equal its solubility as long as the contaminant excess continues. The solubility-leaching flux, therefore, is initially lower than that for the soil-desorption-controlled case, and the curve is flat as long as the concentration in the leaching solution is solubility controlled. Initially, more contaminant remains in the source zone for the solubility-controlled case than for the soil-desorption-controlled case. The leaching release from the solubility-controlled case, therefore, eventually becomes greater than that from the soil-desorption-controlled case.

3.5 Transport and Exposure

Long-term, lifetime exposure analyses evaluated impacts to individuals near the TRU waste locations and to the offsite populations within 80 km (50 miles) for 10,000 years following the loss of institutional control. Radionuclides, organics, and heavy metals were assumed to be released and transported via the atmospheric, groundwater, and surfacewater pathways. The MEPAS code (Section 3.1.2) was used for all transport and exposure calculations. The following sections present the transport and exposure parameters and assumptions used to calculate impacts to individuals and populations.

3.5.1 Transport Parameters

Parameters related to atmospheric release, transport, and deposition analyses require surface-soil characteristics and regional climatological information. The site-specific surface-soil and local climatological information required to estimate soil-suspension rates were obtained from Holdren et al. (1995). The regional meteorological data and atmospheric dispersion data, in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability, were obtained from DOE (1995a). The regional climatology (joint-frequency distributions) and population for regional air receptors for the seven major generator/storage sites are based on Bergenback et al. (1995).

Parameters related to the hydrologic and geologic characteristics of each site analyzed were selected from site-specific environmental settings, also in Holdren et al. (1995). The environmental settings assumed for the location of TRU waste at each major generator/storage site analyzed were presented in Section 3.2. An environmental settings database was used to obtain the number of vadose zone layers at each site, the thicknesses and hydraulic conductivity properties of the vadose zone and aquifer layers, and a suite of physicochemical properties for all layers.

MEPAS source-term calculations involve using site-specific parameters from the environmental setting to calculate secondary parameters that are important to contaminant-release calculations. Table 3.4 shows MEPAS-calculated water-infiltration- and soil-erosion-rate values for each site. The calculated water-infiltration rate is based on local climatology, such as precipitation, cloudiness, wind speed, and humidity; surface-soil properties; and vegetation cover. A water-balance analysis was conducted to estimate the runoff, evapotranspiration, and infiltration rates specific to the site. Table 3.4 estimates the amount of surface soil removed over the 10,000-year modeling period to determine whether the 1.2-m (4-ft) overburden could be removed to expose buried TRU waste to the surface.

Table 3.4. Estimated Water-Infiltration and Soil-Removal Rates for Each Site

	Hanford 200E, 200W	INEEL	LLNL	LANL	ORNL	RFETS	SRS
Infiltration (cm/yr)	1.49	1.43	9.28	0.663	42.2	0.156	24
Annual precipitation (cm)	16	22	37	45	139	39	110
% of precipitation to infiltration (%)	9	6	25	1.4	32	0.4	22
Wind-suspension erosion rate (cm/yr)	5.8E-04	4.5E-02	6.1E-03	1.3E-03	3.5E-05	7.7E-03	2.0E-03
Soil eroded by wind in 10,000 years (ft)	0.19	1.48	2.00	0.44	0.01	2.53	6.59
Overland-flow erosion rate (cm/yr)	1.9E-04	3.4E-03	1.4E-02	5.2E-03	1.2E-02	6.5E-04	1.1E-02
Soil eroded by water in 10,000 years (ft)	0.06	1.11	4.06	1.70	3.79	0.21	3.45
Total soil erosion in 10,000 years (ft)	0.25	1.59	6.06	2.14	3.81	2.74	4.10

3.5.2 Exposure Scenarios

As noted in Section 3.2, TRU waste at the generator/storage sites may be in either buried or surface storage configurations. For analysis of impacts from buried TRU waste, CH- and RH-TRU waste at each site were combined into a single waste-disposal unit, and groundwater, groundwater-to-surfacewater, and air pathways were considered for buried and surface waste units. Contaminants would be leached and released to underlying soils and aquifer systems at depth. Eventually contaminants would reach groundwater and migrate laterally to a downgradient receptor location. Contaminants also eventually may be discharged into nearby surfacewater bodies. Once in these surfacewater systems, dilute concentrations of the contaminants would become available to public water intakes for nearby communities. Contaminants also would suspend (if the cover was eroded away) and volatilize (organic contaminants only) in the air and potentially impact surrounding individuals and populations.

For analysis of impacts from TRU waste left on the surface, CH- and RH-TRU waste at each site were combined into a single waste-storage unit and the groundwater, surfacewater, and air pathways were considered. TRU waste stored in surface facilities would degrade and become available in the environment by the processes of direct air and water erosion, deposition onto soils surrounding the site, and resuspension of contaminated soils in air. The surrounding offsite population would be exposed to these contaminants as they are redistributed into the environment by these cyclic and ongoing processes. Like buried waste, contaminants would be leached and released to underlying soils and aquifer

systems at depth. Eventually, contaminants would reach groundwater and migrate laterally to a down-gradient receptor location. Contaminants eventually could be discharged into nearby surfacewater bodies. Once in these surfacewater systems, dilute concentrations of the contaminants would become available to public water intakes for nearby communities.

The exposure pathways for individual and population impacts were defined differently based on current land use assumptions. Impacts to individuals were evaluated for exposure to contaminants in air and groundwater. The surfacewater pathway was not evaluated because the groundwater flow recharges to the surfacewater pathway where it is further diluted. Therefore, the groundwater pathway for individual impacts would always be higher than from the surfacewater.

Individual Population

The individual air pathway impacts (MEI based on 50th percentile parameter values) evaluated an adult farmer located 100 m from the source. This adult farmer is assumed to live in the same location for 70 years and eat crops from the farm. Inhalation, ingestion, and external dose (from radionuclides only) exposure routes were evaluated.

The individual groundwater impacts from use of contaminated groundwater were calculated for a hypothetical adult farmer assumed to live 300 m downgradient of the waste-storage area. The farmer was assumed to grow and consume his own crops and livestock, using contaminated groundwater as a source of drinking water and for watering the crops and animals. The farmer would be exposed via ingestion of food crops grown in the contaminated soil, inhalation of resuspended contamination, inadvertent ingestion of contaminated soil, and from external radiation from radionuclides in the soil.

Offsite Population

Impacts to the offsite population were evaluated for exposure to contaminants in air and surface-water. Because there is no direct consumption of groundwater by populations around the sites based on current land use, groundwater was not considered to be a significant source of offsite drinking water at the sites, and was considered a transport pathway to surfacewater but not an exposure pathway for populations.

Population impacts from atmospheric releases for all seven sites were calculated using site-specific joint-frequency and population data. The atmospheric population distributions were obtained from DOE (1995b). The air exposure pathways evaluated were inhalations, ingestion of agricultural products (meat, milk, and crops), and external dose from volatilized and suspended contaminants.

Population impacts from domestic and agricultural water uses were considered only for Hanford, ORNL, and SRS because only these sites potentially could release contaminants to surfacewater used by surrounding populations. Information on the populations potentially exposed through water use was

Table 3.5. Site-Specific Population Data Used for Long-Term Lifetime Exposure Analyses

Major Generator/Storage Sites	Population[a] Within 50 mi (80 km)	Population using Surfacewater
Hanford	377,645	33,000[b]
INEEL	153,061	NA
LLNL	6,324,234	NA
LANL	296,546	NA
ORNL	881,652	6,841[c]
RFETS	2,171,877	NA
SRS	620,618	50,000[d]
Total for all sites	10,825,633	89,841

(a) All population distribution data were obtained from the WM PEIS (DOE 1995b).
(b) Data from "Hanford Site Environmental Report for Calendar Year 1994" (Dirkes and Hanf 1995).
(c) Data from "Oak Ridge Reservation Annual Site Environmental Report for 1993" (Martin Marietta Energy Systems, Inc. 1994).
(d) Data from "Savannah River Site Environmental Report for 1992" (Arnett et al. 1995).

gathered by reviewing annual site environmental reports (Arnett et al. 1995; Dirkes and Hanf 1995; Martin Marietta Energy Systems, Inc. 1994) from which specific assumptions for population exposure for each site were made.

3.6 Impacts of Long-Term, Lifetime Exposure

This section provides the results of the long-term lifetime exposure analysis conducted for the SEIS-II No Action Alternative 2. The section is divided into results for the radionuclide, carcinogenic chemical, and noncarcinogenic chemical impacts to individual and populations associated with the seven generator/storage sites. Impacts at each site are those combined from both RH-TRU and CH-TRU waste in surface and buried locations, where applicable.

3.6.1 Radiological Impacts

The estimated radiological impacts for the maximum 70-year lifetime over the 10,000 years following loss of institutional control from release of radionuclides from buried and surface-stored waste at the seven generator/storage sites are presented in Table 3.6. The maximum lifetime MEI dose for all sites ranged from 5E-03 to 7.8 rem. The estimated lifetime probability of an LCF from these doses ranges from 2.6E-06 to 3.9E-03. The highest radiological impacts to an MEI were estimated to occur

Table 3.6. Radiological Impacts to the MEI and Surrounding Populations at the Seven Major Generator/Storage Sites after Loss of Institutional Control

Generator/ Storage Site	Midpoint of Time Period of Maximum Exposure	Lifetime Dose (rem)	Lifetime Probability of LCF	Dominant Exposure Pathway	Major Radionuclide Contributors
Maximally Exposed Individual					
Hanford	35	0.3	1.2E-04	Inhalation	Plutonium-238 Americium-241
INEEL	1,575	7.8	3.9E-03	Groundwater ingestion	Carbon-14
LLNL	35	0.01	6.9E-06	Inhalation	Americium-241 Plutonium-240
LANL	665	0.09	4.5E-05	Inhalation	Plutonium-239 Americium-241
ORNL	105	0.005	2.6E-06	Groundwater ingestion	Plutonium-240
RFEST	35	4.9	2.5E-03	Inhalation	Americium-241
SRS	35	1.4	6.5E-04	Groundwater ingestion	Plutonium-239 Americium-241

Population

Generator/ Storage Site	Midpoint of Time Period of Maximum Exposure	70-Year Population Dose (person-rem)	Number of LCFs	Dominant Exposure Pathway	Major Radionuclide Contributors
Hanford	35	1.4	6.8E-04	Resuspended Soil Ingestion	Plutonium-238 Plutonium-240 Americium-241
INEEL	35	149	0.07	Inhalation	Plutonium-238 Americium-241
LLNL	35	30	0.02	Inhalation	Americium-241 Plutonium-238
LANL	35	162	0.08	Inhalation	Plutonium-239 Americium-241
ORNL	35	0.07	3.6E-05	Inhalation	Americium-241 Plutonium-238
RFETS	665	14,200	7.1	Inhalation	Plutonium-238 Plutonium-240 Americium-241
SRS	9,905	175	0.09	Inhalation	Plutonium-239 Plutonium-240

at INEEL, with groundwater ingestion the dominant exposure pathway and carbon-14 the major radionuclide contributor. These impacts would occur during the 23rd lifetime after loss of institutional control, over 1,500 years after loss of institutional control.

Radiological impacts to the surrounding populations were dominated by inhalation of suspended surface soil. The estimated maximum lifetime population dose ranged from 0.07 to 14,200 person-rem, with the resultant number of LCFs ranging from 3.6E-05 to 7.1. RFETS had the highest estimated number of LCFs, 7.1, occurring during the ninth 70-year period more that 600 years after loss of institutional control. The timing of the maximum population impact is a result of the assumption of the catastrophic failure of the cement waste form at 500 years. The major radionuclide contributors were Pu-238, Pu-240, and Am-241. RFETS was the only site that had an estimated number of LCFs greater than 1. The other sites had estimated LCFs at least 100 times lower than estimates for the RFETS.

The aggregate radiological impacts to the populations at all sites over 10,000 years (approximately 142 70-year lifetimes) were estimated to be 794 LCFs for the No Action Alternative 2 Basic Inventory. These estimates are presented in Table 3.7. This number of LCFs would occur in a total population of about 1,500,000,000 (1.5 billion) people from all seven sites over the 10,000-year analysis period. The population around RFETS would have 781 LCFs, about 98% of the total number of LCFs. Table 3.7 also presents estimates for the Additional Inventory. The Additional Inventory was not analyzed in the WIPP SEIS-II No Action Alternative 2 but was analyzed for Action Alternatives 1, 2,

Table 3.7. Aggregate Radiological Impacts in Populations over 10,000 Years for Seven Major Generator-Storage Sites after Loss of Institutional Control under No Action Alternative 2

Site	Aggregate Impacts from Basic Inventory (LCFs)	Aggregate Impacts from Additional Inventory[a] (LCFs)	Total Aggregate Impacts (LCFs)
Hanford	1 E-02	1 E-02	2 E-02
INEEL	3.8	7.7	11.4
LANL	8.5	5.6	14.1
LLNL	0.4	NA	0.4
ORNL	6 E-04	3 E-04	9 E-04
RFETS	781	NA	781
SRS	0.3	0.1	0.4
Total	794	13	807·
(a) The Additional Inventory was not part of No Action Alternative 2 but was included in estimates of aggregate radiological impacts. NA = no Additional Inventory present.			

and 3. This inventory includes TRU waste, mainly buried waste, that would remain at the sites under No Action Alternative 2 and could result in an additional 13 LCFs in the surrounding populations, for a total of 807 for the two inventories. There is no Additional Inventory at RFETS or LLNL, so the estimated number of LCFs do not change for these two sites.

3.6.2 Carcinogenic Impacts from Hazardous Chemicals

The maximum lifetime probability of cancer incidence to the MEI from exposure to chemical carcinogens ranged from 1.3E-07 to 5.4E-03. Results are presented in Table 3.8. At all sites, the key contaminant was 1,1,2,2-tetrachloroethane, and groundwater ingestion was the major exposure pathway except at LANL. The similarity in key contaminants and exposure pathways is likely a result of the limited inventory information available for hazardous chemicals in TRU waste. The highest probability of cancer incidence was estimated for the MEI at INEEL, with the highest probability of cancer incidence estimated to occur more than 3,000 years after the loss of institutional control, during the 45th, 70-year lifetime.

The maximum lifetime cancer incidence for populations around the sites was estimated to range from 1.1E-07 to 2.8E-04 cancers, also shown in Table 3.8. The highest estimated population cancer incidence was estimated to occur at RFETS during the 70-year period about 500 years after loss of institutional control. The aggregate cancer incidence for all seven sites over 10,000 years was estimated to be 0.002. Figures 3.13 and 3.14 show site-specific hazardous chemical cancer probability and incidence for the MEI and exposed population, respectively, for 70-year periods over the 10,000-year evaluation period.

3.6.3 Noncarcinogenic Impacts from Hazardous Chemicals

Noncarcinogenic impacts from exposure to hazardous chemicals were limited, with most site impacts estimated to occur from exposure to contaminants in groundwater. Noncarcinogenic impacts are presented in terms of a Hazard Index (HI) to indicate potential adverse health effects (non cancer) to an individual. Noncarcinogenic impacts are not calculated for populations because these impacts occur only after a threshold has been exceeded. The HI is the ratio of the estimated daily dose divided by the reference daily dose. An HI of 1.0 is considered the threshold of occurrence for noncarcinogenic effects, and an HI greater than 1.0 is considered a potential concern.

The calculated HIs ranged from 6.4 E-04 to 1.5. Only one site, SRS, had a calculated HI greater than 1.0, with an estimated HI of 1.5 as shown in Table 3.9. The SRS impacts were estimated to occur more than 700 years after the loss of institutional control, from ingestion of contaminated groundwater. The key contaminant was mercury. Figure 3.15 shows the noncarcinogenic impact measure (HI) estimated to occur during each 70-year lifetime at each site for the MEI.

Table 3.8. Hazardous Chemical Carcinogenic Impacts to the MEI and Surrounding Populations at the Seven Major Generator/Storage Sites after Loss of Institutional Control

Generator/ Storage Site	Midpoint of Time Period of Maximum Exposure	Lifetime Probability of Cancer Incidence	Dominant Exposure Pathway	Key Contaminants
Maximally Exposed Individual				
Hanford	1,855	2.3E-06	Groundwater ingestion	1,1,2,2-Tetrachloroethane
INEEL	3,185	5.4E-03	Groundwater ingestion	1,1,2,2-Tetrachloroethane
LLNL	105	5.0E-06	Groundwater ingestion	1,1,2,2-Tetrachloroethane
LANL	525	1.3E-07	Resuspended Soil ingestion	1,1,2,2-Tetrachloroethane
ORNL	35	5.9E-07	Groundwater ingestion	1,1,2,2-Tetrachloroethane
RFETS	2,975	3.1E-07	Groundwater ingestion	1,1,2,2-Tetrachloroethane
SRS	35	3.7E-04	Groundwater ingestion	1,1,2,2-Tetrachloroethane

Generator/ Storage Site	Midpoint of Time Period of Maximum Exposure	Number of Cancers over 70 years	Dominant Exposure Pathway	Key Contaminants
Population				
Hanford	105	1.5E-06	Surfacewater ingestion	1,1,2,2-Tetrachloroethane
INEEL	525	2.9E-06	Resuspended Soil ingestion	1,1,2,2-Tetrachloroethane
LLNL	665	1.1E-07	Resuspended Soil ingestion	Beryllium Cadmium
LANL	525	2.4E-04	Resuspended Soil ingestion	1,1,2,2-Tetrachloroethane
ORNL	665	2.7E-07	Surfacewater ingestion	1,1,2,2-Tetrachloroethane
RFETS	525	2.8E-04	Resuspended Soil ingestion	1,1,2,2-Tetrachloroethane
SRS	665	5.6E-05	Surfacewater ingestion	1,1,2,2-Tetrachloroethane
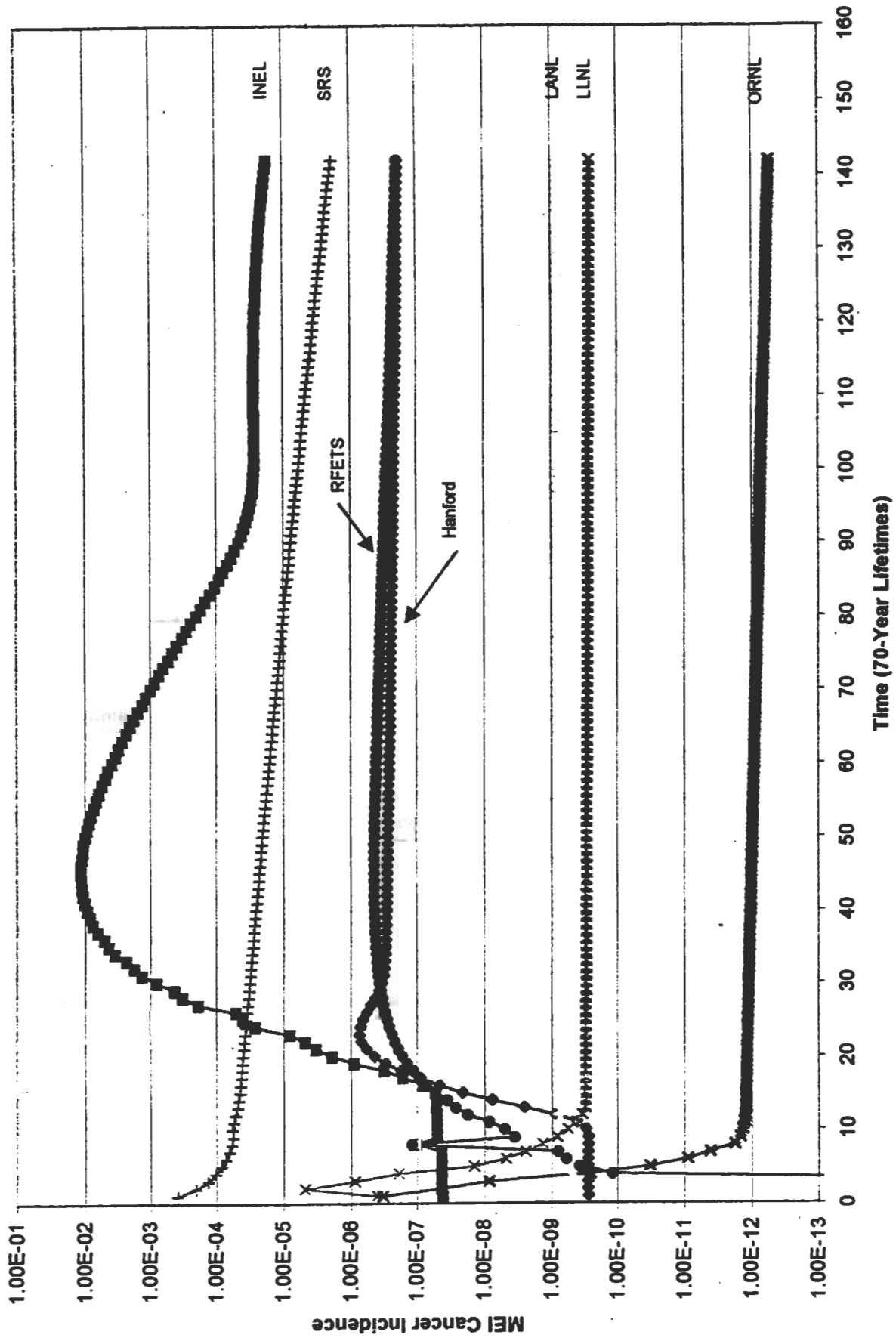
Figure 3.13. Summary of Site-Specific Chemical Carcinogenic Cancer Incidence for the Maximally Exposed Individual

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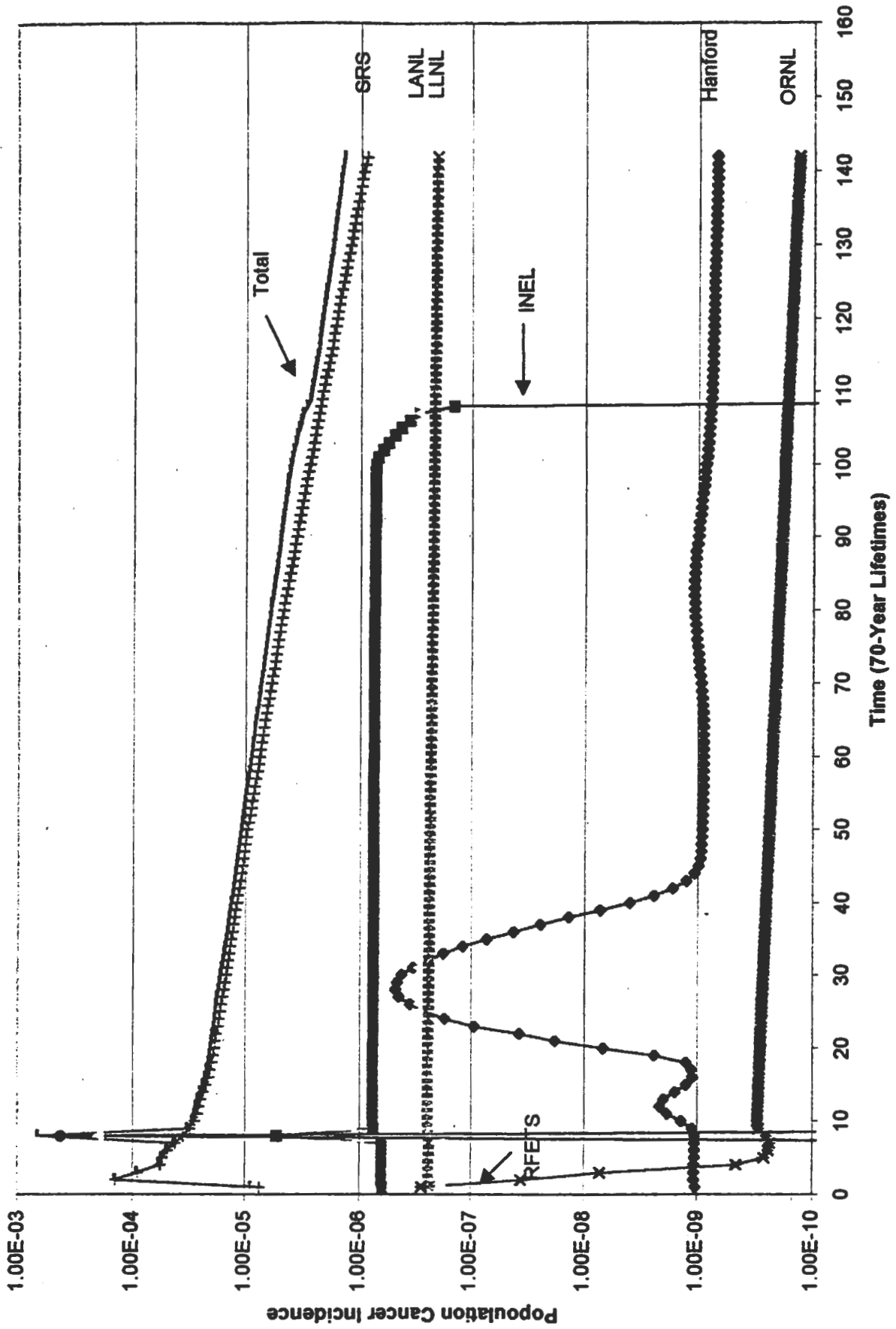
Figure 3.14. Summary of Site-Specific Chemical Carcinogenic Cancer Incidence for Exposed Populations

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Table 3.9. Noncarcinogenic Impacts to the Maximally Exposed Individual at the Seven Major Generator/Storage Sites for the Maximum 70-Year Lifetime

Major Generator/ Storage Site	Midpoint of Time Period of Maximum Exposure	Maximum Hazard Quotient	Dominant Pathway	Hazardous Chemical
Hanford	805	0.2	Groundwater ingestion	Mercury
INEEL	7,035	0.3	Groundwater ingestion	Carbon tetrachloride
LLNL	315	5.3E-04	Groundwater ingestion	Carbon tetrachloride
LANL	665	1.7E-03	Resuspended soil ingestion	Mercury
ORNL	1,295	2.8E-04	Groundwater ingestion	Mercury
RFETS	875	6.4E-05	Groundwater ingestion	Carbon tetrachloride
SRS	735	1.5	Groundwater Ingestion	Mercury
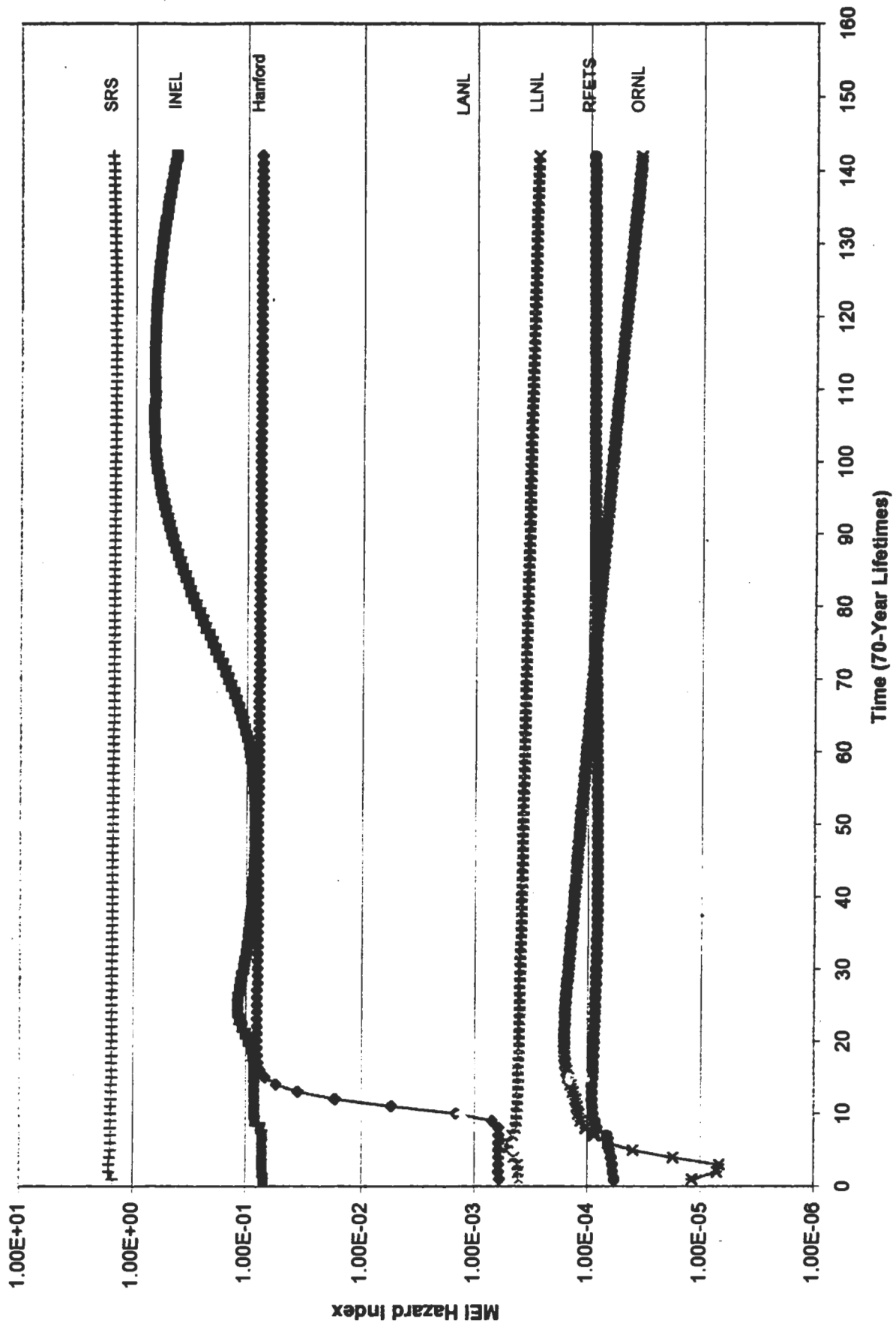
Figure 3.15. Summary of Site-Specific Chemical Noncarcinogenic for the Maximally Exposed Individual

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Appendix A

Contaminant Inventory Data

Appendix A

Contaminant Inventory Data

This appendix contains tables of contaminant inventory data for each site. These tables were obtained from WIPP-SEIS II (DOE 1997) and are broken down by contaminant type (radioactive, organic, and heavy metal) and waste type (contact handled and remote handled). Organic contaminant head-space concentrations and heavy metal contaminant concentrations also are presented.

To model the health effects associated with TRU waste at various sites, the waste must be characterized in terms of its volume, amount of contaminant, and form. This appendix describes the different TRU waste forms modeled and the methods used to determine TRU waste volumes and contaminant inventories for each waste form at the different generator/storage sites.

A.1 Introduction

The TRU wastes analyzed in the FEIS (DOE 1980) and SEIS-I (DOE 1990) included wastes placed in retrievable storage. TRU wastes expected to be generated by ongoing defense activities and programs also were analyzed. The No Action Alternative 2 analysis also needed to deal with waste in retrievable storage and those expected to be generated, but major changes in information and assumptions regarding TRU waste have occurred since the completion of SEIS-I. These changes include:

- Additional sites that have, or expect to generate, TRU waste have been identified. TRU waste volumes at these sites, however, account for only a small percentage of the total TRU waste volume.

- Estimates of the future generation of TRU waste differ for some sites because of changes in mission.

- More detailed descriptions of the volume and physical characteristics of waste streams have been developed at many sites.

- More detailed descriptions of the radionuclide content of waste streams have been developed at many sites.

The information for the updated waste characteristics were taken from four main data sources: DOE 1994, 1995a, 1995b, and 1996.

The database in *Baseline Inventory Report, Revision 2* (BIR-2) (DOE 1995b) contains detailed physical descriptions of TRU waste, as well as TRU waste volumes and radionuclide inventories associated with individual waste streams at generator/storage sites. The database contains both stored TRU waste volumes and TRU waste volumes expected to be generated through the year 2022.

Where possible, BIR-2 (DOE 1995b) was used as the information source for the radionuclide inventories. However, the primary source of radionuclide inventory data was that reported in DOE (1994).

The inventory of hazardous constituents was estimated from DOE (1995a, 1995b, 1996).

Table A.1. Decayed Radionuclide Inventories (Ci) Adjusted for Decay Through 2133 for CH-TRU Waste by Generator/Storage Site for No Action Alternative 2

Isotope	Hanford	INEEL	LANL	LLNL	ORNL	RFETS	SRS
Pu-238	1.71E+05	3.26E+04	9.88E+04	1.80E+02	2.94E+03	5.17E+03	4.99E+05
Pu-239	1.23E+05	4.25E+04	1.51E+05	8.50E+02	3.90E+03	2.67E+05	1.87E+04
Am-241	2.41E+04	8.27E+04	1.90E+04	8.81E+02	3.82E+03	2.25E+05	9.93E+03
Pu-240	2.85E+04	1.01E+04	1.98E+02	3.31E+02	1.27E+03	1.07E+05	4.57E+03
Pu-241	1.52E+03	1.27E+03	2.58E+01	6.87E+01	5.06E+02	1.23E+04	1.82E+03
Pu-242	1.78E+00	7.91E+00	9.23E+02	1.05E-01	1.46E+00	9.11E-04	7.50E-01
U-233	3.74E+02	9.16E+02	8.52E+01	1.10E-06	2.29E+02	1.23E+02	7.50E+00
U-234	3.26E+02	2.18E+01	5.43E+01	9.48E-02	2.16E+01	2.28E+00	2.67E+02
Cs-137	3.20E+02	1.05E+01	9.23E+00	8.63E-07	2.86E-01	0.00E+00	1.50E+00
Y-90	2.97E+02	3.09E+00	7.81E+00	0.00E+00	1.74E+02	0.00E+00	1.28E+00
Sr-90	2.97E+02	3.09E+00	7.81E+00	0.00E+00	1.74E+02	0.00E+00	1.28E+00
Cm-245	7.80E+01	2.38E-01	3.96E-03	0.00E+00	3.96E-02	0.00E+00	0.00E+00
Cm-244	6.95E+00	1.17E+01	6.47E+00	7.36E+00	2.97E+01	0.00E+00	5.06E+01
Th-234	2.74E+01	2.03E-01	4.64E-02	1.57E-01	6.89E-02	0.00E+00	1.14E-02
U-238	2.74E+01	2.03E-01	4.64E-02	1.57E-01	6.89E-02	0.00E+00	1.14E-02
Tc-99	4.45E-05	2.57E-01	2.37E-02	0.00E+00	2.28E+01	0.00E+00	8.99E-06
Pa-233	1.27E+00	1.79E+00	7.69E-02	2.44E-03	1.15E+00	1.61E-01	1.72E+01
Np-237	1.27E+00	1.79E+00	7.69E-02	2.44E-03	1.15E+00	1.61E-01	1.72E+01
Np-239	4.18E-01	6.31E-01	7.21E+00	1.26E-01	1.48E+01	0.00E+00	1.50E+00
Am-243	4.18E-01	6.31E-01	7.21E+00	1.26E-01	1.48E+01	0.00E+00	1.50E+00

Isotope	Hanford	INEEL	LANL	LLNL	ORNL	RFETS	SRS
Th-229	4.12E+00	1.02E+01	9.53E-01	5.31E-09	2.42E+00	1.16E+00	7.06E-02
Ra-225	4.12E+00	1.01E+01	9.53E-01	5.31E-09	2.42E+00	1.16E+00	7.05E-02
Ac-225	4.12E+00	1.01E+01	9.53E-01	5.30E-09	2.42E+00	1.16E+00	7.05E-02
Pb-212	4.55E-17	1.01E+01	3.65E-03	0.00E+00	1.70E-01	0.00E+00	7.02E-02
Ra-224	4.55E-17	1.01E+01	3.65E-03	0.00E+00	1.70E-01	0.00E+00	7.02E-02
Th-228	4.53E-17	1.01E+01	3.65E-03	0.00E+00	1.70E-01	0.00E+00	7.02E-02
U-232	0.00E+00	9.81E+00	3.55E-03	0.00E+00	1.65E-01	0.00E+00	6.84E-02
Ra-226	5.56E-03	1.53E-01	1.65E+00	1.13E-06	8.03E+00	1.81E-05	2.66E-03
Rn-222	1.42E-04	1.52E-01	1.65E+00	1.99E-07	8.03E+00	0.00E+00	1.40E-05
U-235	8.00E+00	9.51E-02	1.00E+00	3.08E-03	2.26E-02	4.52E-04	1.17E-02
Th-231	8.00E+00	9.51E-02	1.00E+00	3.08E-03	2.26E-02	4.52E-04	1.17E-02
Pb-210	1.38E-04	1.50E-01	1.62E+00	1.92E-07	7.84E+00	0.00E+00	1.36E-05
Bi-210	1.38E-04	1.50E-01	1.62E+00	1.92E-07	7.84E+00	0.00E+00	1.36E-05
Po-210	1.38E-04	1.49E-01	1.62E+00	1.92E-07	7.84E+00	0.00E+00	1.36E-05
C-14	7.40E+00	1.90E-01	4.15E-04	0.00E+00	4.14E-03	0.00E+00	0.00E+00

Table A.2. Decayed Radionuclide Inventories (Ci) Adjusted for Decay Through 2133 for RH-TRU Waste by Generator/Storage Site for No Action Alternative 2

Isotope	Hanford	INEEL	LANL	ORNL
Cs-137	1.02E+05	1.18E+03	3.27E+01	1.55E+03
Y-90	8.64E+04	2.25E+03	2.75E+01	4.75E+03
Sr-90	8.64E+04	2.25E+03	2.75E+01	4.75E+03
Pu-239	4.87E+04	3.22E+02	2.24E+02	2.22E+02
Am-241	4.37E+04	3.29E+02	0.00E+00	3.54E+02
Pu-240	2.40E+04	1.26E+02	0.00E+00	3.89E+01
Pu-241	5.52E+03	2.05E+01	0.00E+00	6.82E+00
Pu-238	3.08E+03	6.80E+01	4.28E+00	2.96E+01
Th-231	2.13E+01	9.74E+01	2.13E-02	2.65E+02
U-235	2.13E+01	9.74E+01	2.13E-02	2.65E+02
U-234	1.89E+02	9.38E-01	1.88E-03	2.74E-01
U-233	6.04E+01	3.08E+01	0.00E+00	8.17E+01

Isotope	Hanford	INEEL	LANL	ORNL
Cm-244	0.00E+00	1.26E+01	0.00E+00	2.97E+01
Eu-152	0.00E+00	1.06E+01	6.80E-06	2.88E+01
Cm-243	0.00E+00	6.81E+00	0.00E+00	1.85E+01
C-14	0.00E+00	3.22E+00	0.00E+00	8.61E+00
U-238	1.49E+00	1.78E+00	4.84E-05	4.80E+00
Th-234	1.49E+00	1.78E+00	4.84E-05	4.80E+00
Ni-63	0.00E+00	2.60E+00	0.00E+00	3.03E-01

Table A.3. Volume-Weighted Average Concentrations (ppmv) of Volatile Organic Compounds in CH-TRU Waste for No Action Alternative 2

Volatile Organic Compound	Hanford	LANL	INEEL	SRS	RFETS	ORNL	LLNL	NTS	Mound	ANL-E
Carbon tetrachloride	60.20	271.20	372.68	122.89	274.93	90.19	152.06	93.09	22.44	28.93
Chlorobenzene	9.65	6.20	9.41	8.33	2.67	9.37	9.45	9.39	3.75	12.16
Chloroform	11.84	13.65	17.73	16.13	8.98	18.81	18.53	18.83	3.20	7.35
Methyl ethyl ketone	40.40	21.00	54.23	53.43	19.57	61.45	62.88	61.55	11.22	36.64
Methylene chloride	964.63	753.36	480.29	157.99	91.73	141.75	133.50	141.92	532.07	1,734.17
1,1,2,2-Tetrachloroethane	309.63	128.16	452.87	613.68	141.52	705.13	692.38	706.71	41.00	128.39
Toluene	6.05	3.78	7.04	6.69	2.23	7.57	7.68	7.59	2.02	6.49
1.1-Dichloroethene	8.60	4.47	10.16	12.51	3.21	14.28	13.99	14.31	2.18	7.03
1,2-Dichloroethane	6.01	3.64	6.96	6.64	2.18	7.55	7.66	- 7.56	2.00	6.45
1,1,1-Trichloroethane	22.31	15.67	23.18	21.56	10.12	23.88	24.44	23.95	8.32	26.42

Table A.4. Volume-Weighted Average Concentrations (ppmv) of Volatile Organic Compounds in RH-TRU Waste for No Action Alternative 2

Volatile Organic Compound	Hanford	LANL	INEEL	ORNL
Carbon tetrachloride	26.13	212.58	119.03	172.05
Chlorobenzene	12.61	8.97	8.80	6.53
Chloroform	10.08	20.34	17.44	12.58
Methyl ethyl ketone	43.76	29.16	56.88	42.19
Methylene chloride	1,586.14	1,161.64	172.75	94.56
1,1,2,2-Tetrachloroethane	241.67	145.58	638.56	500.94
Toluene	7.17	5.20	7.05	5.35
1.1-Dichloroethene	8.87	5.89	13.07	10.12
1,2-Dichloroethane	7.13	5.11	7.02	5.26
1,1,1-Trichloroethane	28.01	21.05	22.56	17.69

Table A.5. Concentration of Hazardous Metals

Metal	CH-TRU Waste Inventory (kg/m³)	RH-TRU Waste Inventory (kg/m³)
Beryllim	1.21E-01	1.21E-01
Cadmium	1.73E-03	1.74E-03
Lead	4.79E+00	4.64E+02
Mercury	2.05E+00	2.05E+00

Table A.6. Inventory (g) of Volatile Organic Compounds in CH-TRU Waste for the No Action Alternative 2

Compound	Hanford	INEEL	LANL	ORNL	RFETS	SRS	LLNL
Carbon tetrachloride	139,660	438,499	230,351	6,730	120,509	59,830	7,333
Chloroform	20,560	15,608	8,674	1,050	2,946	5,878	669
1.1-Dichloroethene	3,759	2,253	715	201	265	1,147	127
Methylene chloride	698,580	176,405	199,743	3,302	12,551	24,010	2,010
1,1,2,2-Tetrachloroethane	16,234,213	12,042,561	2,460,196	1,189,188	1,401,930	6,752,290	754,557
Chlorobenzene	216,336	106,937	50,878	6,756	11,324	39,169	4,401
Methyl ethyl ketone	136,670	93,043	26,013	6,686	12,509	37,925	4,421
Toluene	56,419	33,311	12,894	2,272	3,939	13,090	1,489
1,2-Dichloroethane	24,633	14,463	5,461	994	1,691	5,705	652
1,1,1-Trichloroethane	47,554	25,053	12,230	1,637	4,074	9,645	1,083

Table A.7. Inventory (g) of Volatile Organic Compounds in RH-TRU Waste for the No Action Alternative 2

Compound	Hanford	INEEL	LANL	ORNL
Carbon tetrachloride	31,024	9,397	1,959	25,450
Chloroform	8,958	1,031	140	1,392
1.1-Dichloroethene	1,983	194	10	282
Methylene chloride	587,915	4,257	3,342	4,366
1,1,2,2-Tetrachloroethane	6,485,477	1,139,327	30,327	1,674,680
Chlorobenzene	144,586	6,712	798	9,323
Methyl ethyl ketone	75,757	6,547	392	9,100
Toluene	34,223	2,238	193	3,181
1,2-Dichloroethane	14,949	978	83	1,374
1,1,1-Trichloroethane	30,552	1,636	178	2,405

Table A.8. CH- and RH-TRU Waste Inventories for No Action Alternative 2 by Site

Site	Additional Inventory for CH-TRU (m^3)	Additional Inventory for RH-TRU (m^3)
Hanford	63,000	1,000
INEEL	57,000	440
LLNL	0	0
LANL	14,000	120
ORNL	260	2,000
RFETS	0	0
SRS	4,900	0
Total	139,163	3,563

A.2 References

U.S. Department of Energy (DOE). 1994. *Integrated Data Base Report-1994: U.S. Spent Nuclear Fuel and Radioactive Waste Inventories, Projections, and Characteristics*. DOE/RW-0006, Revision 11, Oak Ridge National Laboratory, Oak Ridge, Tennessee.

U.S. Department of Energy (DOE). 1995a. *Waste Isolation Pilot Plant Safety Analysis Report*. DOE/WIPP-95-2065, Revision 1, November, Carlsbad, New Mexico.

U.S. Department of Energy (DOE). 1995b. *Draft. Waste Management Programmatic Environmental Impact Statement for Managing Treatment, Storage, and Disposal of Radioactive and Hazardous Wastes*. DOE/EIS-0200, Office of Environmental Management, Washington, D.C.

U.S. Department of Energy (DOE). 1996. *Comment Responses and Revisions to the Resource Conservation and Recovery Act Part B Permit Application*. DOE/WIPP 91-005, Revision 5.2, January, Carlsbad, New Mexico.

Appendix B

Contaminant Release and Groundwater Transport Properties for Long-Term, Lifetime Exposure Analysis

Appendix B

Contaminant Release and Groundwater Transport Properties for Long-Term, Lifetime Exposure Analysis

Various calculations made in the intruder and long-term impact analysis require contaminant property data as input variables. Degradation/decay coefficients and sorption K_d values are needed to determine flux factors and unit transport factors. Solubilities are needed to determine flux factors. Effective diffusion coefficients in cement are needed to determine flux factors for waste forms in the cemented waste-form category. This appendix describes the methods, including computer codes, used to obtain specific values for these parameters at the different sites and tabulates the values used in instances where these values were not "default" values reported in referenced literature.

B.1 Contaminant Solubilities

Modeling solubility bounded by contaminant release from all waste forms depends on contaminant/waste-form/site-specific solubility limits. The MEPAS chemical database (augmented from that reported in Strenge and Peterson [1989]) contains solubility values for organic chemicals but has little solubility information for radionuclides and metals. This is because solubility values for inorganic contaminants depend on many site-specific properties, and it is difficult to supply a single meaningful point value for these contaminants. Developing site-specific solubilities for all contaminants in the site inventory would be a formidable, and ultimately unnecessary, task. Instead, a screening was done to determine which contaminants would be the primary contributors to impacts at each site, initially ignoring solubility concerns. If contaminants are present in low concentrations, have low toxicity values, decay rapidly, and/or have very low mobility, they could be eliminated from further consideration without generating a solubility limit. The screened list was further reduced by calculating the highest aqueous-phase concentration for each contaminant at each site, based on soil properties and contaminant K_d values. If the highest possible water concentration was <0.001 mg/L, it was assumed that solubility would not be important enough to merit calculation. The analysis focused on radioactive contaminants. However, lead was also included in the analysis because it was known to exist in large quantities because of its use as a shielding material.

The solubility calculations were completed for 6 of the 7 sites under consideration. Calculations were not completed for the LLNL because it was added to the list of sites at a later time and by that time, the solubility calculations had been shown to have only minor effects on risk calculations. The solubilities were calculated for two possible waste forms: soil/debris and cement. Solubility limits for cemented waste forms stored at the Hanford Site were not calculated because the Hanford Site does not currently have any of this waste form. Solubility limits for cemented waste forms at the SRS also were not calculated because concerns about this waste form were established later in the analysis after the solubility calculations had shown to have only minor effects on the risk calculation. Waterborne

contaminants of concern that had possible water concentrations of 0.001 mg/L included lead, neptunium, plutonium, and uranium. This effort attempted to calculate conservative maximum concentration limits for each of these contaminants, based on equilibrium solubility considerations. The following subsections describe how the inorganic solubilities used in the analyses were calculated.

B.1.1 Geochemical Speciation Code and Associated Thermodynamic Databases

The site-specific inorganic solubilities were calculated using the chemical equilibria code MINTEQA2 (Version 3.11), which is based on equilibrium thermodynamic solubility principles. See Garrels and Christ (1965), Lindsay (1979), Morel (1983), Nordstrom and Munoz (1985), Stumm and Morgan (1981), and Allison et al. (1991) regarding the use of thermodynamic principles to calculate solubility equilibria in aqueous systems.

As part of a PNNL study, the database obtained with the MINTEQA2 code was expanded to include the aqueous species and solids containing the radionuclide elements americium, neptunium, plutonium, radium, technetium, thorium, and uranium. These database modifications and the sources of these data are described in detail in Krupka and Serne (1996). The database changes for the aqueous species and solids containing uranium are based on the compilation by Grenthe et al. (1992) and supersede those listed in the MINTEQA2 database as obtained from EPA.

B.1.2 Geochemical Environments

Prior to running the MINTEQ computer code, various assumptions must be made regarding specific geochemical conditions. The geochemical environments selected for modeling represent a best estimate, based on the resources available for the solubility-modeling calculations. Three geochemical environments were considered for the solubility calculations. Each environment conceptually corresponds to a leachate that results from the interaction of soil-pore water with each type of waste form (i.e., soil/debris and cement) as a closed environment under equilibrium conditions. These conditions should result in conservative maximum concentration limits, in that the concentrations of contaminants will be decreased by dilution in open flow-through systems.

In the absence of site-specific waste-form leachate data it was assumed that, except for pH and concentration of dissolved carbonate, the leachate composition would mimic that of the soil-pore water. Assumptions regarding the oxidation/reduction conditions also were required to model the geochemical environment. Site-specific pore-water compositions, oxidation/reduction conditions, and conceptual models used for determining the pH and concentration of dissolved carbonate for each waste form are discussed below.

The compositions selected to simulate soil-pore waters at each site are listed in Table B.1. These data were compiled from the following sources:

- Hanford Site - Serne et al. (1993)

Table B.1. Compositions of Soil-Pore Waters Used for Solubility Modeling

	Hanford	INEEL	LANL	ORNL	RFETS	SRS
pH	7.9	7.68	7.9	7.3	See LANL	4.9
Constituent, mg/L						
Na	35	258	14	3.3	See LANL	10
K	6.8	3.6	2.1	3.1	See LANL	<0.1[a]
Ca	58	65.6	18	39.2	See LANL	8.6
Mg	15	32.6	4.4	4.5	See LANL	<0.1[a]
H_4SiO_4	63.6	NL[b]	70.4	16.2	See LANL	NL
HCO_3	34	695.6	77	142	See LANL	<0.05[a]
SO_4	85	186.4	9	8	See LANL	30
Cl	71	59.9	15	1.6	See LANL	4
F	1	NL	1.7	NL	See LANL	NL
NO_3	NL	NL	NL	NL	See LANL	8
PO_4	NL	NL	NL	NL	See LANL	NL

(a) Limiting value used for solubility calculations.
(b) NL = None listed.

- Idaho National Engineering Laboratory - Personal communication, January 1996, C. A. Dicke (Senior Scientist, Idaho National Engineering Laboratory, Lockheed Martin Idaho Technologies, Idaho Falls, Idaho)

- Los Alamos National Laboratory - Environmental Protection Group (1994)

- Oak Ridge National Laboratory - Davis et al. (1984)

- Savannah River Site - Savannah River Laboratory (1985).

A source for soil-pore compositions could not be identified for the RFETS. Based on the approach used for other aspects of this risk analysis (e.g., K_d sorption values), it was assumed that composition of soil-pore water for this site could be approximated by that used for the LANL.

Waste storage has assumed to be in surface or near-surface soil environments, where there is free exchange of oxygen and carbon dioxide gas between the atmosphere, soil, pore waters, and waste form. Because of kinetic considerations and formation of corrosion products, it was also assumed that the corrosion of any metal constituents (e.g., metallic iron) in the waste would have no impact on the oxidation state of the leachate. As a result of these two assumptions, the solubilities were calculated using only the most oxidized valence state of environmental consequence for each contaminant. In most cases, the solubilities of the reduced valence states of the contaminants of concern are expected to be less than those calculated for the oxidized forms.

B.1.2.1 Soil/Debris-Waste Leachate Compositions

The leachate from soil/debris waste was assumed to have the same composition as the site pore water listed in Table B.1. Because the waste is expected to contain a significant amount of site soil, the leachate composition is expected to be controlled by the same reactions that control the composition of the soil-pore water. The leachate pH also was assumed to be the same as that listed for the soil-pore water.

Because carbonate is a strong complexant of some dissolved contaminants, such as uranium, the concentrations of dissolved carbonate used for the soil/debris leachate were modeled in two ways. One set of calculations was based on the concentration of dissolved carbonate listed for the soil-pore waters in Table B.1. A second set of calculations assumed that the concentration of dissolved carbonate was fixed by assumed equilibrium with carbon dioxide gas. For this latter calculation, a partial pressure of 0.003 atm carbon dioxide (10 times that of atmospheric carbon dioxide) was used to constrain the concentration of dissolved carbonate. This pressure is often used as the reference for carbon dioxide in soil systems (Lindsay 1979). To provide the most conservative maximum concentration limits for soil/debris waste, Table B.2 only lists the larger of the two contaminant-solubility concentrations calculated from these different conceptual models for dissolved carbonate.

B.1.2.2 Cemented Waste Leachate Composition

Except for pH and dissolved carbonate concentrations, the leachate from cemented waste was assumed to have the same composition as the site pore water listed in Table B.1.

The pH of the cement leachate was assumed to be 12.5 as a result of buffering by the solubility of portlandite, $Ca(OH)_2$, in cement. In the absence of detailed reactive transport computer simulations of how the cement leachate pH changes as a function of time from dissolution of cemented waste forms at site-specific environmental factors (e.g., water-recharge rates), this pH value was considered a conservative, limiting pH value for cement leachate. The chemical reactions associated with the hydration of cement are described in detail in International Atomic Energy Agency (1993), Atkins and Glasser (1992), and Reardon (1992).

The concentrations of dissolved carbonate were fixed by assuming equilibrium with the solubility of calcite, $CaCO_3$. Calcite is known to precipitate as a result of carbonation reactions occurring with the dissolution of cement (Criscenti and Serne 1990; Duerden et al. 1990; Reardon and Dewaele 1990; Smith and Walton 1991).

B.4

Table B.2. Calculated Solubilities Used for Long-Term, Lifetime Exposure Analysis (milligrams/liter)

Contaminant	Hanford 200E Soil/Debris	Hanford 200W Soil/Debris	INEEL Soil/Debris	INEEL Cement	LANL Soil/Debris	LANL Cement	ORNL Soil/Debris	ORNL Cement	RFETS Soil/Debris	RFETS Cement	SRS Soil/Debris	SRS Cement	LLNL Soil/Debris	LLNL Cement
^{14}C	<0.001[a]	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
^{243}Cm	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
^{244}Cm	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Pb	0.2[c]	0.2[c]	0.2[c]	40,000[c]	0.2[c]	40,000[c]	0.4[c]	40,000[c]	0.2[c]	40,000[c]	2,000[c]	NC[c]	NC	NC
^{237}Np	900[c]	900[c]	2,000[c]	10[c]	900[c]	50[c]	2,000[c]	50[c]	900[c]	50[c]	700,000[c]	NC	NC	NC
^{238}Pu	70[c]	70[c]	500[c]	0.00005[c]	50[c]	0.000005[c]	20[c]	0.000005[c]	50[c]	0.00001[c]	0.001[c]	NC	NC	NC
^{239}Pu	70[c]	70[c]	500[c]	0.00005[c]	50[c]	0.000005[c]	20[c]	0.000005[c]	50[c]	0.0001[c]	0.001[c]	NC	NC	NC
^{240}Pu	70[c]	70[c]	500[c]	0.00005[c]	50[c]	0.000005[c]	20[c]	0.000005[c]	50[c]	0.0001[c]	0.001[c]	NC	NC	NC
^{233}U	200[c]	200[c]	500[c]	0.001[c]	200[c]	0.9[c]	50[c]	5[c]	200[c]	0.9[c]	7[c]	NC	NC	NC
^{234}U	200[c]	200[c]	500[c]	0.001[c]	200[c]	0.9[c]	50[c]	5[c]	200[c]	0.9[c]	7[c]	NC	NC	NC
^{235}U	200[c]	200[c]	500[c]	0.001[c]	200[c]	0.9[c]	50[c]	5[c]	200[c]	0.9[c]	7[c]	NC	NC	NC

(a) NC = Not calculated.
(b) <0.001 - Maximum aqueous concentration is <0.001 mg/L.
(c) Calculated solubility, initially solubility limited.
(d) Calculated solubility, not solubility limited.

B.5

B.1.3 Solid-Phase Solubility Controls

Maximum concentration limits for dissolved lead, neptunium, plutonium, and uranium were calculated using the MINTEQA2 code. The concentration limits are based on solubility controls selected for each of the dissolved contaminants, based on the contaminant-containing solid phases in the MINTEQA2 thermodynamic database.

Selection of the most appropriate solubility control(s) was based on phase-stability information given in published studies and the experience that project staff have with the geochemistry of radio-nuclide aqueous systems. The selected solubility controls are the following:

- lead - cerussite ($PbCO_3$) for leachates from all waste types

- neptunium - $NpO_2(OH)(am)$ for leachates from all waste types

- plutonium - $PuO_2H_2O(am)$ for leachates from all waste types

- uranium - schoepite [$UO_2(OH)_2H_2O$] for leachates from soil/debris and uranophane [$Ca(H_3O)_2(UO_2)_2(SiO_4)_23H_2O$] for cement leachate.

Except for lead, the rationale for selecting the solubility controls used for the other contami-nants listed above is discussed by Krupka and Serne (1996). They used geochemical modeling to simulate radionuclide solubilities over the pH range from 4.0 to 12.5 in support of a performance assessment of a hypothetical low-level waste-disposal facility. Cerussite was selected for the solubility control of lead, based on analyses by Lindsay (1979) of chemical equilibria in soil/water systems. The solubilities based on the amorphous solids are expected to be highly conservative maximum concentration limits. These solids are known to precipitate at laboratory time scales but will crystallize over time to solids that are more thermodynamically stable and have lower solubili-ties. The kinetic rates for completing these crystallization processes are not known for radionuclide solids in soil/water environments.

For other contaminants, such as uranium, modeling and/or laboratory evidence suggests that the solubilities of different solid phases may be controlling the concentrations of their constituent metals over different pH regions. For example, two solubility controls, schoepite and uranophane, were considered in calculating maximum concentration limits of dissolved uranium. Schoepite is known to precipitate readily in low-temperature aqueous systems at laboratory time scales and result in high concentrations of dissolved uranium (Krupka et al. 1985). In natural low-temperature aqueous systems, the presence of alkali and/or alkaline earth ions at high pH conditions would result in the precipitation of alkali/alkaline earth uranium compounds that would control the solu-bility of uranium to concentrations lower than those resulting from equilibrium with schoepite. Concentration limits, based on schoepite, are, therefore, suspected to be highly conservative. Atkins et al. (1990, 1991) investigated uranium reactions with portlandite and hydrated calcium silicate hydrogel solutions. Three uranium-containing phases, which included uranophane, a

hydrated calcium uranyl oxide [$Ca_2UO_5(1.2-1.5)H_2O$], and becquerelite ($CaU_6O_{19}H_2O$) were identified by X-ray diffraction and analytical electron microscopy in the resulting mixtures. The solubilities of these phases will, therefore, represent more realistic concentration limits for dissolved uranium in cementitious systems than those based on the solubility of schoepite. However, the solubilities of the hydrated calcium uranyl oxide phase and becquerelite cannot be calculated because the required thermodynamic data for these phases are not available.

B.1.4 Estimated Contaminant Solubilities

During the screening analysis, expected aqueous concentrations (in the absence of solubility limitations) of each contaminant of concern were calculated, based on inventory, K_d, and volume of pore water in the waste zone. If the maximum concentration was below 0.001 mg/L, it was assumed that solubility would not be a controlling factor in determining the contaminant's aqueous concentration. Table B.2 shows if solubility is an important issue for each contaminant/waste form/site combination. Table B.2 also reports the calculated solubility values for the remaining combinations where solubility limitations may be an issue. These calculated solubility values were compared to the maximum aqueous concentrations determined in the screening analysis to determine which contaminant/waste form/site combinations would be solubility limited. Table B.2 showed that plutonium, uranium, and lead are the only contaminants that have releases that are controlled by solubility at one or more sites.

A few of the reported solubilities did not affect the analysis but required further comment regarding their accuracy. The maximum concentration limit of 700,000 mg/L, calculated for neptunium in pH 4.9 leachate from soil/debris waste at the SRS, is considered unrealistic. It is suspected that the adequacy of the thermodynamic data for neptunium, especially solubility-controlling phases, may be a contributing factor in the high concentrations predicted for the solubility of neptunium. However, the maximum water concentration calculated for neptunium-237 at the SRS, based on contaminant K_d and soil properties, is 0.23 mg/L. Therefore, it is likely that a more realistic solubility value would still exceed the 0.23-mg/L value, and that the release of neptunium-237 is not controlled by solubility.

The maximum concentration limit calculated for lead in cemented waste leachates at pH 12.5 is 40,000 mg/L. This value is also considered to be unrealistically high. The unrealistic solubility value calculated for lead probably results from inadequate thermodynamic data for the lead hydrolysis species that are predicted to be present at the high pH value for cemented leachate. The maximum lead leachate concentrations from ORNL and LANL, based on K_d and soil properties, are 300 and 200 mg/L, respectively. These K_d-controlled concentrations also are thought to be much higher than expected. However, the high-solubility limit was used in this analysis to determine if lead accounts for a large portion of the risk. These conservative numbers did not result in high risk numbers for lead, so refinements to the solubility limits were not required.

B.2 Contaminant-Sorption Coefficients

The MEPAS chemical database (augmented from that reported in Strenge and Peterson [1989] by adding more contaminants and updating some parameter values) contains information needed to calculate contaminant-sorption coefficients from the physical and chemical properties of the soil. For organic chemicals, the database tabulates values of the carbon matter partition coefficient and octanol-water partition coefficient for each contaminant. The K_d value can be calulated from either of these values, using the organic carbon content of the soil (and a correlation equation if the octanol-water partition coefficient is used). For contaminants other than organic chemicals, the database contains a set of K_d values for each contaminant that varies, depending on the soil pH and total weight percent of clay, organic matter, and iron and aluminum oxyhydroxides. If information needed to calculate a K_d value for a specific contaminant was not available in the MEPAS database, the contaminant was initially assigned a conservative value of zero. K_d values calculated from these MEPAS database parameters/methods are assumed as default values unless better site-specific information was available.

The MEPAS database parameters were used directly to compute K_d values for the organic contaminants. However, an attempt was made to improve the validity of the MEPAS inorganic K_d values by finding site-specific K_d values for the various sites under study. Adsorption phenomenology for inorganic contaminants is not well understood, based on first principles; the most acceptable approach to quantifying adsorption is to use site-specific studies as opposed to generic literature, when possible. Site geochemists were contacted at the sites under study; several documents were obtained that contain site-specific K_d values for some contaminants, and, in a few cases, unpublished information was used on site preferences for K_d values for near-surface soils and natural waters (the closest scenario to expected leachate from a solid-waste-burial ground for all but cement-stabilized wastes). The site-specific references consulted were Brookins (1984), Davis et al. (1984), Dickie (personal communication), Hietanen et al. (1985), Kaplan and Serne (1995), Kaplan et al. (1995), Looney et al. (1987), Serne et al. (1993), Sheppard et al. (1984), Sheppard and Thibault (1990), Triay et al. (1994), and Wolfsberg (1980). In addition to the information received from the site, some chemical analog assumptions were invoked to generate less-conservative (i.e., nonzero) K_d values for rare elements that are chemically quite similar to elements for which K_d data were available (e.g., curium is similar to americium, praseodymium is similar to europium, etc).

These site-specific and chemical analog K_d values were used to create an alternative lookup table to compare with the MEPAS database values. Site-specific K_d values were used when they were available, and the chemical analog K_d values were used instead of conservative zero values. If neither of these two values was available, the MEPAS database values were used. If neither site-specific, chemical analog, nor MEPAS database values were available, a conservative zero value was used in the calculations. The final radionuclide and inorganic K_d values for each soil type for each environmental setting are shown in Table B.3.

Table B.3. K_d Values Used in Long-Term, Lifetime Exposure Analyses (milliliters of water/gram of soil)

Contaminant	Hanford, All Zones, Both Settings	ORNL, PZ1, PZ3, SZ	ORNL, PZ2	ORNL, PZ4	LANL, PZ1	LANL, PZ2, PZ3, SZ	SRS, All Zones	INEEL, PZ1, PZ3	INEEL, PZ2, PZ4, SZ	Rocky PZ	Rocky SZ	LLNL, All Zones
Lead	1000	597	10	12.1	25	25	100	597	234	1830	234	597
Mercury	322	580	60	500	5280	580	10000	580	322	5280	322	580
Beryllium	70	1400	140	800	8000	1400	140	1400	70	8000	70	1400
Cadmium	14.9	423	42.9	100	567	423	6.3	423	14.9	567	14.9	423
AC225	228	538	6	6	4600	538	6	538	228	4600	228	538
AC227	228	538	6	6	4600	538	6	538	228	4600	228	538
AC228	228	538	6	6	4600	538	6	538	228	4600	228	538
AG108M	10	4	4	6	40	4	4	4	0.4	40	0.4	580
AG110	10	4	4	6	40	4	4	4	0.4	40	0.4	580
AG110M	10	4	4	6	40	4	4	4	0.4	40	0.4	580
AM241	82	5670	200	1000	130	130	100	200	82	1000	82	200
AM243	82	5670	200	1000	130	130	100	200	82	1000	82	200
AM245	82	5670	200	1000	130	130	100	200	82	1000	82	597
AT217	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
BA137M	530	2800	280	1600	946	946	280	2800	530	16000	530	2800
BI210	234	597	10	12	1830	597	10	597	234	1830	234	597
BI211	234	597	10	12	1830	597	10	597	234	1830	234	597
BI212	234	597	10	12	1830	597	10	597	234	1830	234	597
BI213	234	597	10	12	1830	597	10	597	234	1830	234	597
BI214	234	597	10	12	1830	597	10	597	234	1830	234	597
BK249	82	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
BK250	82	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
C14	0	0	0	0	0	0	0.01	0	0	0	0	0
CD109	14.9	423	42.9	100	567	423	6.3	423	14.9	567	14.9	423
CD113M	14.9	423	42.9	100	567	423	6.3	423	14.9	567	14.9	423
CE144	288	538	6	6	4600	538	1000	538	288	4600	288	538
CF249	82	Conservative 0	Conservative 0	Conservative 0	50	50	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
CF250	82	Conservative 0	Conservative 0	Conservative 0	50	50	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
CF251	82	Conservative 0	Conservative 0	Conservative 0	50	50	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
CF252	82	Conservative 0	Conservative 0	Conservative 0	50	50	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
CM242	82	Conservative 0	Conservative 0	Conservative 0	50	50	3000	4000	4000	Conservative 0	Conservative 0	Conservative 0
CM243	82	Conservative 0	Conservative 0	Conservative 0	50	50	3000	4000	4000	Conservative 0	Conservative 0	Conservative 0
CM244	82	Conservative 0	Conservative 0	Conservative 0	50	50	3000	4000	4000	Conservative 0	Conservative 0	Conservative 0

Table B.3. (contd)

Contaminant	Hanford, All Zones, Both Settings	ORNL, PZ1, PZ3, SZ	ORNL, PZ2	ORNL, PZ4	LANL, PZ1	LANL, PZ2, PZ3, SZ	SRS, All Zones	INEEL, PZ1, PZ3	INEEL, PZ2, PZ4, SZ	Rocky PZ	Rocky SZ	LLNL, All Zones
CM245	82	Conservative 0	Conservative 0	Conservative 0	50	50	3000	4000	4000	Conservative 0	Conservative 0	Conservative 0
CM246	82	Conservative 0	Conservative 0	Conservative 0	50	50	3000	4000	4000	Conservative 0	Conservative 0	Conservative 0
CM247	82	Conservative 0	Conservative 0	Conservative 0	50	50	3000	4000	4000	Conservative 0	Conservative 0	Conservative 0
CM248	82	Conservative 0	Conservative 0	Conservative 0	50	50	3000	4000	4000	Conservative 0	Conservative 0	Conservative 0
CO58	100	780	0.9	0.2	200	8.81	10	8.81	1.94	200	1.94	8.81
CO60	100	780	0.9	0.2	200	8.81	10	8.81	1.94	200	1.94	8.81
CS134	300	64000	24.9	27	428	249	501	75	51	270	51	249
CS135	300	64000	24.9	27	428	249	501	75	51	270	51	249
CS137	300	64000	24.9	27	428	249	501	75	51	270	51	249
EU150	228	538	6	6	50	50	6	538	228	4600	228	538
EU152	228	538	6	6	50	50	6	538	228	4600	228	538
EU154	228	538	6	6	50	50	6	538	228	4600	228	538
EU155	228	538	6	6	50	50	6	538	228	4600	228	538
FE55	15	46800	10	12.9	15	15	10	15	15	15	15	15
FE59	15	46800	10	12.9	15	15	10	15	15	15	15	15
FR221	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
FR223	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
H3	0	0	0	0	0	0	0	0	0	0	0	0
I129	0.7	11.7	0	0	0	0	0.2	0	0	0	0	0
MN54	16.5	25.3	2.5	4	36.9	25.3	2.5	25.3	16.5	36.9	16.5	25.3
NB95	50	100	100	100	100	100	100	100	50	100	50	100
NB95M	50	100	100	100	100	100	100	100	50	100	50	100
NI59	12.2	58.6	5.86	65	50	50	100	58.6	12.2	650	12.2	58.6
NI63	12.2	58.6	5.86	65	50	50	100	58.6	12.2	650	12.2	58.6
NP237	2.4	Conservative 0	Conservative 0	Conservative 0	0.001	0.001	10	5	5	Conservative 0	Conservative 0	Conservative 0
NP239	2.4	Conservative 0	Conservative 0	Conservative 0	0.001	0.001	10	5	5	Conservative 0	Conservative 0	Conservative 0
NP240	2.4	Conservative 0	Conservative 0	Conservative 0	0.001	0.001	10	5	5	Conservative 0	Conservative 0	Conservative 0
NP240M	2.4	Conservative 0	Conservative 0	Conservative 0	0.001	0.001	10	5	5	Conservative 0	Conservative 0	Conservative 0
PA231	Conservative 0	Conservative 0	Conservative 0	Conservative 0	100	100	Conservative 0	550	550	Conservative 0	Conservative 0	Conservative 0
PA233	Conservative 0	Conservative 0	Conservative 0	Conservative 0	100	100	Conservative 0	550	550	Conservative 0	Conservative 0	Conservative 0
PA234	Conservative 0	Conservative 0	Conservative 0	Conservative 0	100	100	Conservative 0	550	550	Conservative 0	Conservative 0	Conservative 0
PA234M	Conservative 0	Conservative 0	Conservative 0	Conservative 0	100	100	Conservative 0	550	550	Conservative 0	Conservative 0	Conservative 0
PB209	1000	597	10	12.1	1830	597	10	597	234	1830	234	597
PB210	1000	597	10	12.1	1830	597	10	597	234	1830	234	597

Table B.3. (contd)

Contaminant	Hanford, All Zones, Both Settings	ORNL, PZ1, PZ3, SZ	ORNL, PZ2	ORNL, PZ4	LANL, PZ1	LANL, PZ2, PZ3, SZ	SRS, All Zones	INEEL, PZ1, PZ3	INEEL, PZ1, PZ4, SZ	Rocky PZ	Rocky SZ	LLNL, All Zones
PB211	1000	597	10	12.1	1830	597	10	597	234	1830	234	597
PB212	1000	597	10	12.1	1830	597	10	597	234	1830	234	597
PB214	1000	597	10	12.1	1830	597	10	597	234	1830	234	597
PD107	10	4	4	39.3	50	50	4	4	0.4	40	0.4	4
PM147	10	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
PO210	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PO211	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PO212	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PO213	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PO214	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PO215	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PO216	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PO218	5.9	14.9	14.9	14.9	14.9	14.9	400	14.9	5.9	14.9	5.9	14.9
PR144	10	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	500	500	Conservative 0	Conservative 0	Conservative 0
PU236	80	100	4	43	110	100	100	100	10	250	10	100
PU238	80	100	4	43	110	100	100	100	10	250	10	100
PU239	80	100	4	43	110	100	100	100	10	250	10	100
PU240	80	100	4	43	110	100	100	100	10	250	10	100
PU241	80	100	4	43	110	100	100	100	10	250	10	100
PU242	80	100	4	43	110	100	100	100	10	250	10	100
PU243	80	100	4	43	110	100	100	100	10	250	10	100
PU244	80	100	4	43	110	100	100	100	10	250	10	100
RA223	24.3	100	100	124	200	100	100	100	24.3	124	24.3	100
RA224	24.3	100	100	124	200	100	100	100	24.3	124	24.3	100
RA225	24.3	100	100	124	200	100	100	100	24.3	124	24.3	100
RA226	24.3	100	100	12.4	200	100	100	100	24.3	124	24.3	100
RA228	24.3	100	100	124	200	100	100	100	24.3	124	24.3	100
RH106	10	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
RN219	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
RN220	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
RN222	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0
RU106	27	351	26	26	690	351	158	351	274	690	274	351
SB125	2	6	5	15.9	15.9	6	5	6	2	15.9	2	6
SB126	2	6	5	15.9	15.9	6	5	6	2	15.9	2	6

Table B.3. (contd)

Contaminant	Hanford, All Zone, Both Settings	ORNL, PZ1, PZ3, SZ	ORNL, PZ3	ORNL, PZ4	LANL, PZ1	LANL, PZ4, PZ3, SZ	SRS, All Zones	INEEL, PZ1, PZ3	INEEL, PZ1, PZ4, SZ	Rocky PZ	Rocky SZ	LLNL, All Zones
SE...ord	2	6	5	15.9	15.9	6	5	6	2	15.9	2	6
SE79	0	14.9	14.9	14.9	2	2	2.5	14.9	5.91	14.9	5.91	14.9
SM151	228	538	6	6	50	50	6	538	228	4600	228	538
SN119M	100	10	5	5	50	10	100	10	5	10	5	10
SN121M	100	10	5	5	50	10	100	10	5	10	5	10
SN126	100	10	5	5	50	10	100	10	5	10	5	10
SR90	24.3	494	100	12.4	116	100	7.9	8.3	8.3	124	24.3	100
TC99	0	20	20	20	0.3	0.3	0.001	0.1	0.1	20	3	20
TEI25M	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	125	125	Conservative 0	Conservative 0	Conservative 0
TE127	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	125	125	Conservative 0	Conservative 0	Conservative 0
TE127M	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	Conservative 0	125	125	Conservative 0	Conservative 0	Conservative 0
TH227	100	500	60	100	500	500	100	500	100	2700	100	500
TH228	100	500	60	100	500	500	100	500	100	2700	100	500
TH229	100	500	60	100	500	500	100	500	100	2700	100	500
TH230	100	500	60	100	500	500	100	500	100	2700	100	500
TH231	100	500	60	100	500	500	100	500	100	2700	100	500
TH232	100	500	60	100	500	500	100	500	100	2700	100	500
TH234	100	500	60	100	500	500	100	500	100	2700	100	500
TL207	0	0.2	0	0	0.8	0.2	0	0.2	0	0.8	0	0.2
TL208	0	0.2	0	0	0.8	0.2	0	0.2	0	0.8	0	0.2
TL209	0	0.2	0	0	0.8	0.2	0	0.2	0	0.8	0	0.2
U232	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
U233	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
U234	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
U235	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
U236	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
U237	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
U238	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
U240	0.1	50	5	50	1.8	1.8	40	50	0	500	0	50
Y90	228	538	6	6	4600	538	6	538	228	4600	228	538
ZN65	12.7	939	280	280	1460	939	16	939	12.7	1460	12.7	939

B.3 Contaminant-Diffusion Coefficients in Cement

The methods for modeling contaminant release from cemented waste forms depend on contaminant-specific cement-diffusion coefficients. The cement-diffusion-coefficient values used in the No Action Alternative 2 analyses were obtained from the MEPAS chemical database. This database (augmented from that reported in Serne and Wood [1990]) contains cemented waste-form effective diffusion coefficients for most of the contaminants of concern. These database values were reviewed for consistency, given that new literature has been obtained since its original creation. The only changes made involved improving overly conservative values for curium and neptunium. Curium was given a value similar to more-common actinides by using the concept of chemical analogs that indicates that effective diffusion coefficients of chemically similar elements should be the same. The value for neptunium was changed, based on observations that neptunium does not leach from cements/grouts and, thus, should have an effective diffusion coefficient similar to the other actinides as opposed to a high value like nitrate. Neptunium leach data are found in Ewart et al. (1986, 1992).

If values were not available in the MEPAS chemical database and a chemical analog was not available, the contaminant was assigned a conservatively high value of 5.0E-08 cm^2/s. Organic contaminants are typically not well immobilized by cement and are also assigned a value of 5.0E-08 cm^2/s. The final list of cement-diffusion coefficients used in the analyses is shown in Table B.4.

B.4 References

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Table B.4. Grout-Diffusion Coefficient Used for Long-Term, Lifetime Exposure Analyses

Contaminant	Grout-Diffusion Coeffcient (cm^2/s)
Americium-241	5.0E-13
Americium-243	5.0E-13
Carbon-14	1.0E-12
Curium-243	5.0E-11
Curium-244	5.0E-11
Cesium-137	5.0E-10
Europium-152	5.0E-11
Neptunium-237	5.0E-10
Protactinium-233	Conservative 5.0E-08
Plutonium-238	5.0E-11
Plutonium-239	5.0E-11
Plutonium-240	5.0E-11
Plutonium-241	5.0E-11
Radium-226	5.0E-11
Stontium-90	5.0E-11
Uranium-233	1.0E-12
Uranium-234	1.0E-12
Uranium-235	1.0E-12
Lead	1.0E-11
Beryllium	5.0E-10
Cadmium	5.0E-10
Mercury	5.0E-11
Carbon tetrachloride	Conservative 5.0E-08
Chloroform	Conservative 5.0E-08
Methylene chloride	Conservative 5.0E-08
Carbon disulfide	Conservative 5.0E-08
1,1-Dichloroethylene	Conservative 5.0E-08
Methyl ethyl ketone	Conservative 5.0E-08
1,1,2,2-Tetrachloroethane	Conservative 5.0E-08
Toluene	Conservative 5.0E-08
Chlorobenzene	Conservative 5.0E-08
1,2 Dichloroethane	Conservative 5.0E-08

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Appendix C

Geology and Hydrology of the Generator/Storage Sites

Appendix C

Geology and Hydrology of the Generator/Storage Sites

This appendix describes the geohydrology of the seven major generator/storage sites.

C.1 Hanford Site

Figure C.1 presents stratigraphic columns for each setting. Hydrogeologic data for the various settings are presented in Holdren et al. (1995, Tables 5.2 through 5.8). Surfacewater data used also are presented in Holdren et al. (1995, Table 5.1). A brief description of the hydrogeology of the 200-East and 200-West environmental settings of the Hanford Site are as follows.

The meteorological and atmospheric dispersion data in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability used in this analysis for Hanford are presented in Bergenback et al. (1995).

200-East Area. The water table in the 200-East Area is contained within a number of hydrostratigraphic units (Connelly et al. 1992). As a necessary simplification, the water table was placed at the contact between the Hanford coarse-grained facies and the Ringold Formation, which is true for the central and southern portions of the 200-East Area. Above the water table, sediments are divided into three distinct partially saturated zones: 1) upper Pasco gravels, 2) sandy sequence, and 3) lower Pasco gravels. The suggested thicknesses of these units are 10, 50, and 19 m, respectively.

The unconfined aquifer occurs primarily in the Ringold Formation sediments. The thickness of the unconfined aquifer in the 200-East Area has been reported to be up to 38 m; however, the saturated zone thickness was selected to be 9 m, which is the generally accepted thickness of the contaminated portion of the unconfined aquifer in this area of Hanford.

200-West Area. The water table in the 200-West Area occurs in the Ringold Formation. Reported hydrostratigraphic units above the water table include eolian deposits, Hanford formation coarse-grained facies, Hanford formation slackwater facies, early Palouse soil, the Plio-Pleistocene unit, the upper Ringold unit, and the unsaturated portion of Ringold unit E. The five partially saturated zones selected to represent this environmental setting are the Hanford formation coarse-grained facies, Hanford formation slackwater facies, the Plio-Pleistocene unit, the upper Ringold unit, and the Ringold unit E. The surficial eolian deposit was not included because of a lack of data for these deposits and their variable thicknesses. Because of the possibility that some early Palouse soil samples may have been misidentified, the Hanford formation slackwater beds and early Palouse soil unit were combined.
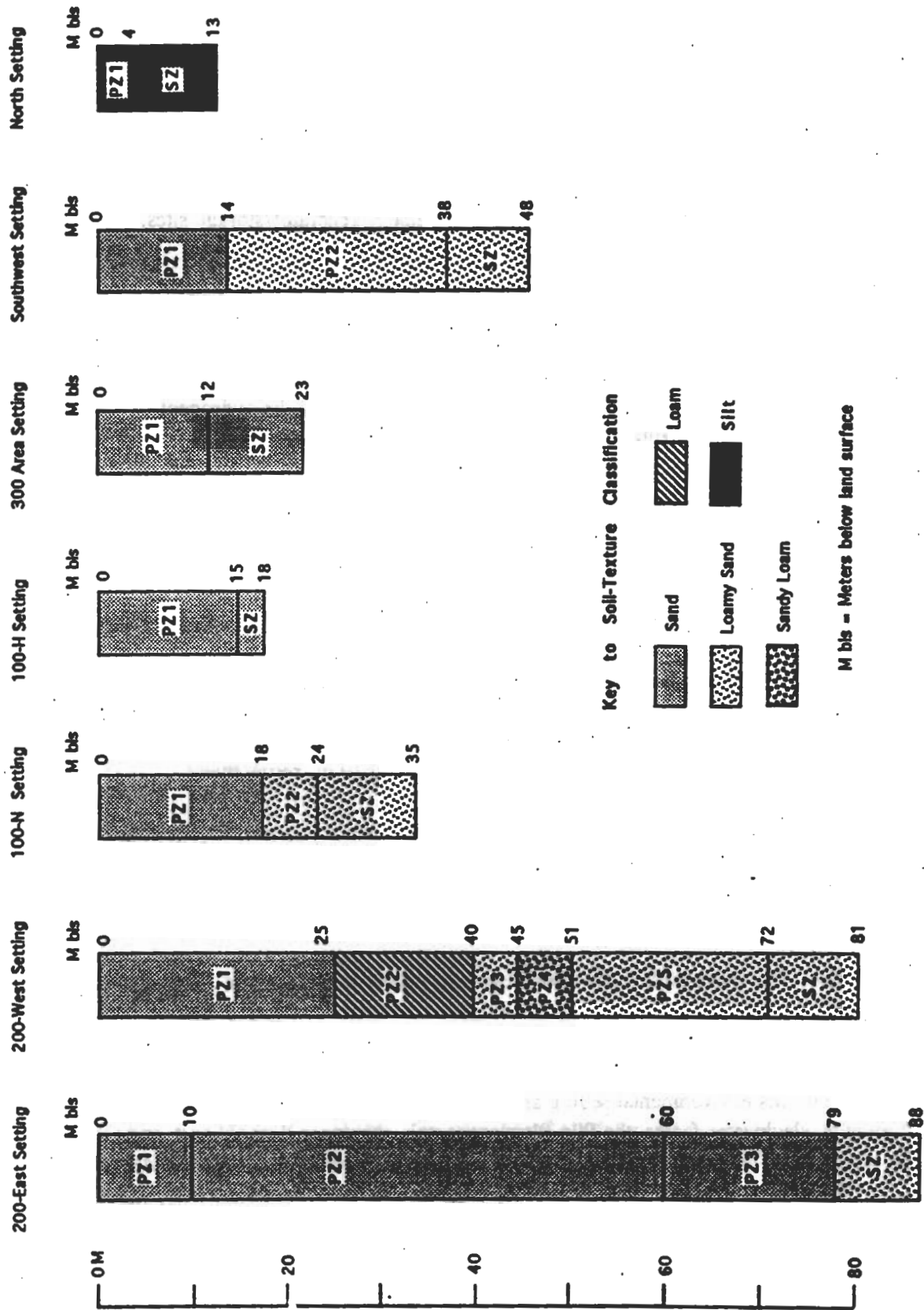
Figure C.1. Stratigraphic Columns for Hanford Site Environmental Settings

C.2

The upper Ringold unit is not continuous throughout the 200-West Area; however, this unit was included because of its potential significant impact on infiltration in the 200-West Area.

In the 200-West Area, the unconfined aquifer occurs in Ringold Formation gravel unit E, except in the northwestern part of the setting, where the aquifer occurs in gravel unit A. The reported thickness of the unconfined aquifer is up to 84 m. However, the contaminated portion of the aquifer is generally accepted to be 9 m, so this thickness was selected to represent the effective aquifer thickness.

C.2 Idaho National Engineering and Environmental Laboratory

The underlying rocks at INEEL include basaltic lava flows interbedded with sediments to depths of several thousand meters. This system of basalt rocks and interbedded sedimentary sequences makes the major groundwater unit beneath the INEEL site referred to as the Snake River Plain Aquifer. Although the general groundwater-flow patterns at the INEEL are understood (Barraclough et al. 1981; Lewis and Jensen 1985), the subsurface is heterogeneous and, as a result, specific flow paths may be complex. This aquifer system is a major water supply in the region and is considered a sole-source aquifer. Water in this aquifer discharges to the Snake River several hundred kilometers downgradient from the site.

Surface drainage at INEEL does not connect to the Snake River (the largest major river in the region). Three river drainages do exist at the INEEL: 1) Big Lost River, which enters INEEL at its southwestern boundary; 2) Little Lost River; and 3) Birch Creek, which enters the site at its northern boundary. During most years, the Big Lost River terminates near the INEEL's boundary in a series of diversion ponds constructed in the 1970s. During episodes of high flow, surfacewater from the Big Lost River moves in the channel past several major facilities to a series of natural playas in INEEL's northcentral portion. During most years, all surfacewaters in the Little Lost River and Birch Creek watersheds are diverted to irrigation before entering the INEEL. During periods of high flow, surfacewaters from these two streams can reach the playas.

Figure C.2 presents stratigraphic columns for each of these settings. Hydrogeologic data for the settings are presented in Holdren et al. (1995, Tables 6.2 through 6.5). A brief description of the hydrogeology of the RWMC setting follows.

The meteorological and atmospheric dispersion data in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability used in this analysis for INEEL are presented in Bergenback et al. (1995).

RWMC Setting. The RWMC setting is located on the southern and western margins of the INEEL. The Big Lost River enters this setting from the west. During most years, river flow terminates in a series of diversion ponds constructed near the southwestern corner of the INEEL. The stratigraphy within the partially saturated zones at the RWMC is complex, consisting of a series of basalt flows and interbedded sediments. As found in other areas within INEEL, basalts in this area tend to be vertical and, thus, allow for the rapid migration of contaminants downward through these zones. A number of
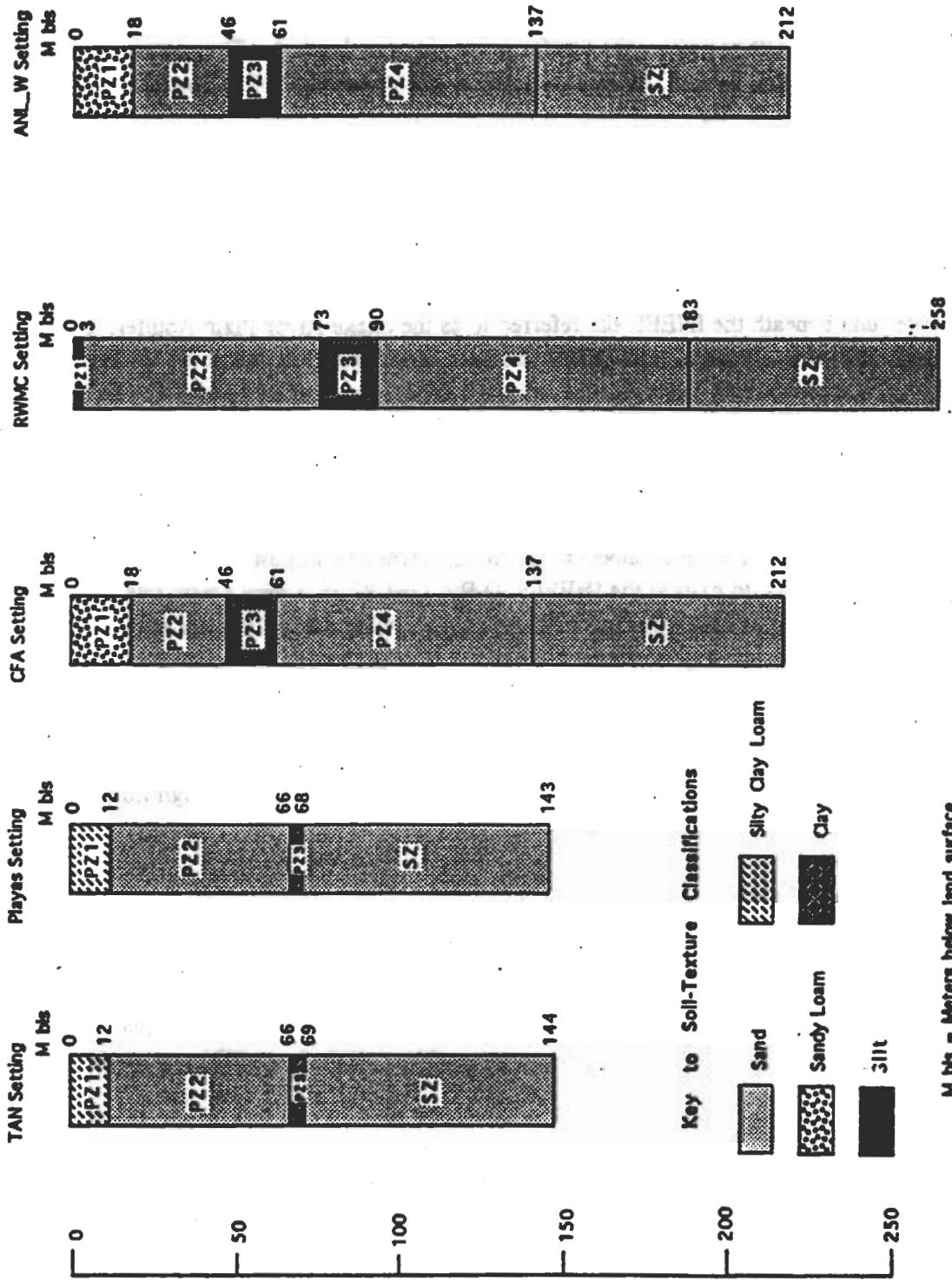
Figure C.2. Stratigraphic Columns for Idaho National Engineering and Environmental Laboratory Environmental Settings

C.4

the sedimentary interbeds are dominated by sands and gravels and, as such, offer little resistance to the migration of contaminants. One sedimentary zone, however, is predominantly fine grained and is capable of significantly retarding the migration of radionuclides. As a result of this distribution of hydraulic characteristics at the RWMC, this setting has been divided into four partially saturated zones and one saturated zone. The first partially saturated zone consists of surface sediments and soils. The second and fourth zones are modeled as vertically fractured basalts, and the third partially saturated zone attempts to describe the fine-grained sedimentary interbed located at a depth of about 64 m. Depth to groundwater in this area is greater than at other areas at INEEL. The total thickness of the partially saturated zone is modeled at about 180 m.

Groundwater moves off the INEEL through its southern boundary. This boundary includes sections of the RWMC, the Central Facilities Area (CFA), and the Argonne National Laboratory-West (ANL-W) hydrogeologic settings. This groundwater is discharged to surfacewaters several hundred kilometers downgradient at springs and seeps along the Snake River.

C.3 Lawrence Livermore National Laboratory

Within the Livermore Valley, valley-fill deposit can be divided into three separate units: 1) Pleistocene- and Holocene-aged alluvium, 2) upper Livermore Formation, and 3) lower Livermore Formation (Sims et al. 1990). The upper Livermore Formation beneath LLNL is made up of complexly interfingered alluvial deposits of highly variable permeability and thickness. The more-permeable deposits vary in thickness from 1 to 10 m. The less-permeable sediments are commonly 1- to 7-m thick and can be as much as 25-m thick. Deposits of the upper Livermore Formation are nearly identical to the overlying Pliestocene- and Holocene-aged alluvial deposits. The permeable parts of the upper and lower Livermore Formations are separated by a horizontally extensive, low-permeability, confining layer that occurs near the upper part of the lower Livermore Formation (Layton et al. 1990).

Groundwater generally is found in two aquifer systems at LLNL: 1) an upper, shallow aquifer system in the alluvium and upper Livermore Formation, and 2) a lower, confined aquifer system within the lower Livermore Formation. Owing to the presence of the low-permeability sediments in the upper 10 to 30 m of the lower Livermore Formation, little or no hydraulic communication is thought to occur between the upper and lower aquifer systems (Layton et al. 1990). Natural recharge occurs primarily along the fringes of the basin and through arroyos during periods of winter flow. Groundwater movement in the valley is toward the central east-west axis of the valley and then westward to the Central Basin (Sims et al. 1990).

No perennial surfacewater can be found near LLNL (Layton et al. 1990). When it exists, it generally drains from the main site west to arroyos and intermittent streams.

Figure C.3 presents the stratigraphic column for the main setting. Hydrogeologic data for the main setting are presented in Holdren et al. (1995, Table 11.1). A brief description of the hydrogeology of this setting is as follows.
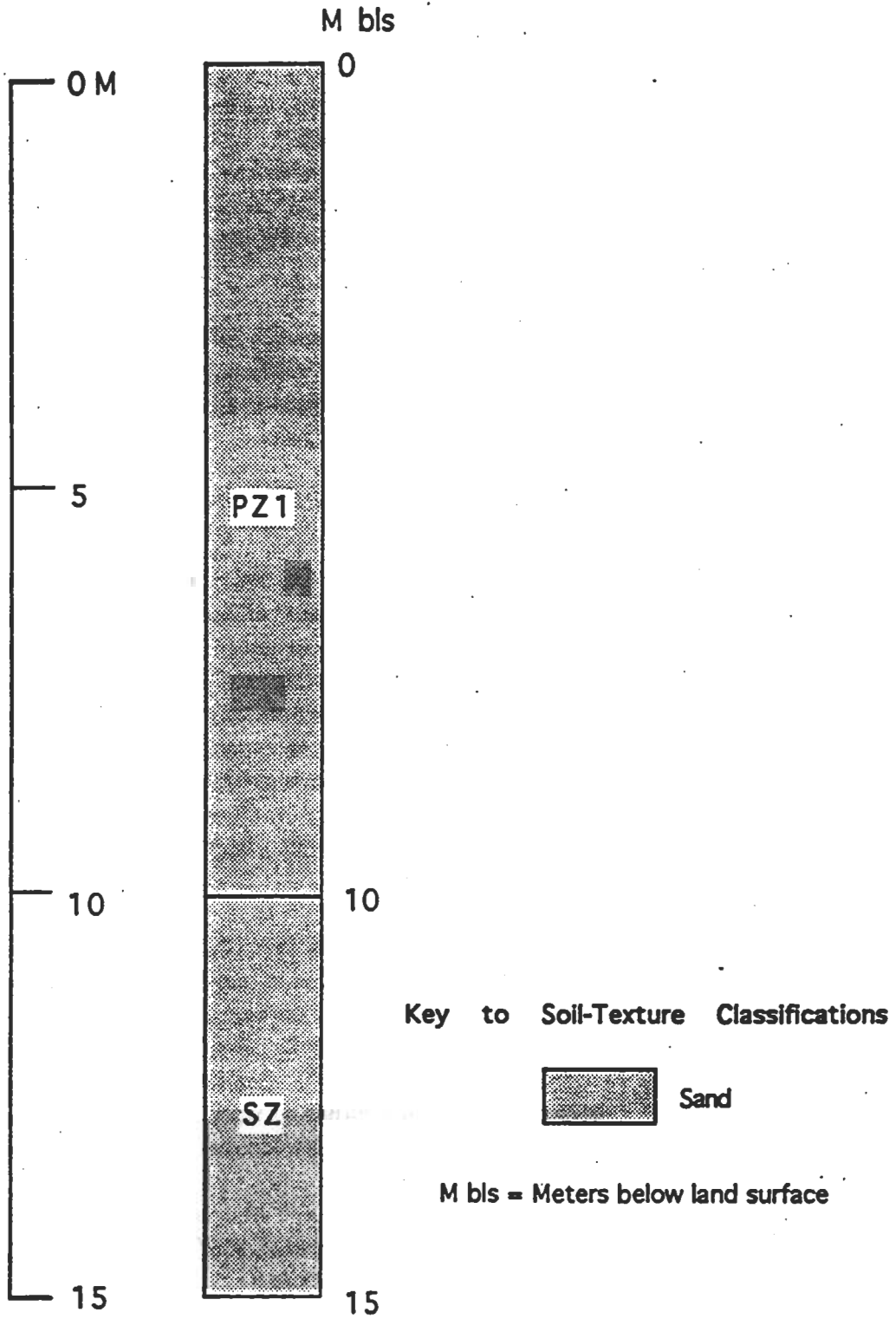
Figure C.3. Stratigraphic Column for Lawrence Livermore National Laboratory Environmental Setting

The main setting is assumed to have one partially saturated zone and one saturated zone as depicted in Figure C.3. Although alluvial sediments of the main setting are quite heterogeneous, the heterogeneity is too fine a scale to be captured by the approach used in this analysis. Input parameters for the unsaturated and saturated zones given in Holdren et al. (1995) were derived from information provided in Layton et al. (1990).

The meteorological and atmospheric dispersion data in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability used in this analysis for the LLNL are presented in Bergenback et al (1995).

C.4 Los Alamos National Laboratory

Major rock units beneath LANL, from oldest to youngest, include Tesuque Formation sediments, Chino Mesa Formation basalts, Puye Formation conglomerates, Tschicoma Formation volcanics, and Bandelier Tuff. Groundwater at LANL occurs within shallow alluvium, perched aquifers found within the rock units identified above. A main regional aquifer is found in the Sante Fe Group sediments, which include parts of the Tesuque and Puye formations.

Groundwater occurs in three modes in the LANL area: 1) water in shallow alluvium in some of the larger canyons, 2) perched water, and 3) main aquifer of the Los Alamos area (LANL 1992). Intermittent and ephemeral stream flows in the canyons of the Pajarito Plateau have deposited alluvium that ranges in thickness to as much as 30 m. Perched water at LANL occurs in conglomerates and basalts beneath the alluvium in a limited area in the mid-reach of the Pueblo Canyon and in a second area near the confluence of lower Pueblo and Los Alamos canyons.

The main aquifer is the only aquifer capable of supplying municipal and industrial water for the LANL area. The upper surface of the main aquifer rises westward from the Rio Grande River through the Tesuque Formation and lower Puye Formation conglomerate beneath the central and western portions of the plateau (DOE 1979; LANL 1992). Surfacewater and groundwater in the alluvium are separated from water in the main aquifer by several hundred meters of unsaturated volcanic tuff and sediments (LANL 1992).

The major surfacewater feature of the LANL area is the Rio Grande River, which flows southward, with a drainage area of about 37,000 km^2 in southern Colorado and northern New Mexico. The eastern margin of the Pajarito Plateau stands about 90 to 275 m above the Rio Grande River (LANL 1992). In the LANL area, there is intermittent stream flow in canyons cut into the Pajarito Plateau. Perennial flow to the Rio Grande River occurs in canyons north and south of LANL.

One of the four hydrogeologic environmental settings used to represent LANL in Holdren et al. (1995). Figure C.4 presents stratigraphic columns for each of these settings. Hydrogeologic data for the various settings are presented in Holdren et al. (1995, Tables 12.1 through 12.4). A brief description of the hydrogeology of the Mesa Tops setting for LANL follows.
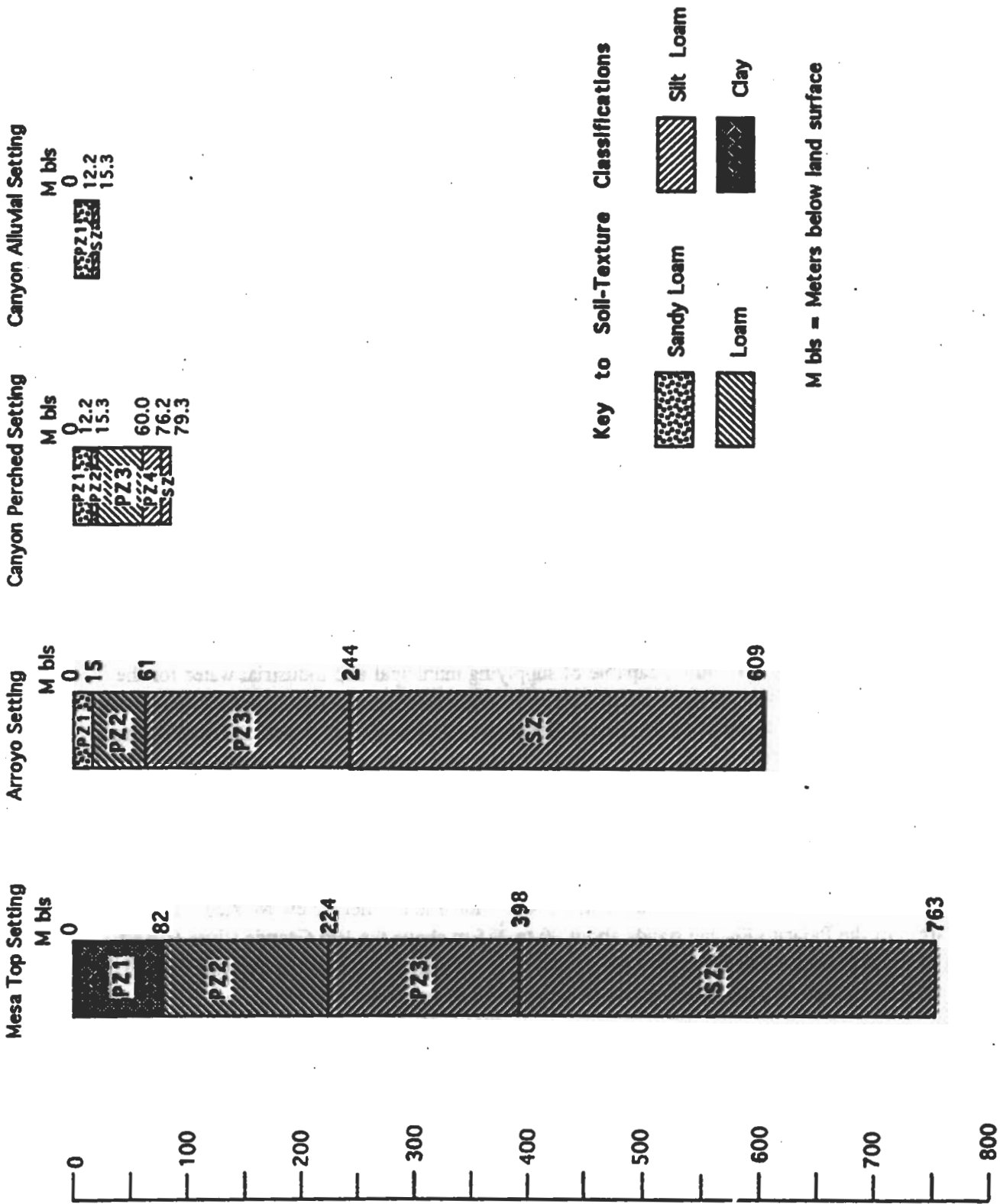
Figure C.4. Stratigraphic Columns for Los Alamos National Laboratory Environmental Settings

C.8

The main aquifer at LANL is encountered at depths either below the previously discussed canyon bottoms or below the various finger-like mesas of the Pajarito Plateau. Mesa tops are capped with Bandelier Tuff and its two members, the Tshirege and the Otowi. Proceeding west to east across the Pajarito Plateau, the thickness of the Bandelier Tuff ranges from 320 to 80 m (DOE 1979). Values for the hydrogeologic parameters used to represent the mesa tops environmental setting are primarily based on data from deep well EGH-LA-1, located in the northcentral portion of the Pajarito Plateau between Mortandad and Sandia canyons (see Figure 3.4). Thickness of the partially saturated zone is about 400 m, based on depth-to-water measurements (LANL 1993). Thickness of the saturated zone is about 370 m, based on an estimated thickness of potable water in the main aquifer (DOE 1979). Hydraulic conductivity estimates and soil-texture classifications were performed as above in the canyon alluvial aquifer, canyon perched aquifer, and canyon bottom settings.

The meteorological and atmospheric dispersion data in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability used in this analysis for LANL are presented in Bergenback et al. (1995).

C.5 Oak Ridge National Laboratory

The Oak Ridge Reservation (ORR) is located in the Bethel and Melton valleys, part of the Tennessee Valley and Ridge Province of the Southern Appalachian fold and thrust belt. The area is characterized by a succession of northeast-trending thrust faults that stack and repeat the Paleozoic section. As a result of thrusting and subsequent differential erosion, a series of ridges, consisting of relatively resistant rocks (e.g., sandstone, shale, and dolomite), and valleys developed in more easily eroded material (e.g., less-resistant carbonate and shale). Most units dip steeply toward the south-southeast. This structure has a major role in regulating groundwater movement in the area. The main area is situated in the Bethel Valley; Chestnut Ridge lies to the north, and Haw Ridge lies to the south. Chestnut Ridge is formed from the Knox Group, which is a deeply weathered cherty dolomite. The Bethel Valley is underlain by the Chickamauga Limestone, which consists of silty limestones and calcareous siltstones. Haw Ridge is formed from the Rome Formation, consisting of sandstones, silt-stones, and shales. The Melton Valley, located south of Haw Ridge, is underlain by the Conasauga Group, which consists of calcareous shales interlayered with limestone and siltstone. The Copper Creek fault lies south of the main area of Haw Ridge. Fracturing is pervasive throughout all rock units at the ORR and also occurs in zones that are associated with faults. Although the rock units are highly fractured, most have been filled with mineral precipitates (predominantly calcite) and do not contribute to a secondary permeability. Karstification, or solution enlargement of existing fractures, is common in the more carbonate-rich units and has been reported from the Knox Group and Chickamauga Group (Martin Marietta Energy Systems, Inc. 1989).

Although groundwater occurs in all formations that crop out at the ORR, three major hydrologic units are present. Mechanisms and rates of flow appear to be controlled by topography, structure, and lithology. The first hydrologic unit is a near-surface storm-flow zone that extends across all of the ORR (as well as other ORR areas). Because of its proximity to the surface, there are no clearly defined, partially saturated zones that overlie the storm-flow zone. The storm-flow zone transmits

90% of the incoming precipitation laterally to surfacewater. The remaining 10% infiltrates to underlying aquifers. The second hydrologic unit is the Knox aquifer, which includes areas underlain by the Knox Group and the Maynardville Limestone Member of the Conasauga Group. The third hydrologic unit is the ORR aquitards, consisting of areas underlain by all other geologic formations. Vertical movement of these waters tends to occur along contact zones between major lithologic units. Therefore, soil waters generally move down (dip) until reaching a zone of saturation, where they then tend to migrate along strike until reaching an area of discharge.

The Clinch River is the major surfacewater source that receives discharges from the ORR. Four Tennessee Valley Authority reservoirs influence the flow and/or water levels of the lower Clinch River: Norris and Melton Hill on the Clinch River and Watts Bar and Fort Loudon on the Tennessee River. The ORR is bounded on the south and west by a 63-km stretch of the Clinch River. Melton Hill Dam is located at Clinch River kilometer 37.2 (river mile 23), forming the Melton Hill Reservoir and several major embayments that bound the ORR. Both groundwater and surfacewater are drained from the ORR by a network of small tributaries of the Clinch River. The ORR drains into White Oak Creek and Melton Branch, which both drain into the Clinch River (Martin Marietta Energy Systems, Inc. 1989).

One of the three hydrogeologic environmental settings used to represent ORNL in Holdren et al. (1995). The storm-flow zone has no partially saturated zones associated with it. The Knox and Aquitard settings share a series of four partially saturated zones, named inceptisol 1 (silt loam), inceptisol 2 (sandy loam), ultisol 1 (silt loam), and ultisol 2 (clay loam). The combined thickness of the four partially saturated zones is 8 m. Figure C.5 presents stratigraphic columns for each setting. Hydrogeologic data for the various settings are presented in Holdren et al. (1995, Tables 17.1 through 17.4). A brief description of the hydrogeology of the Knox setting follows.

In the general hydrologic framework developed by hydrogeologists at ORNL, there are two hydro-stratigraphic units controlling groundwater flow. These are designated the Knox aquifer and the ORR aquitards. The Knox aquifer consists of the Knox Group and the underlying Maynardville Limestone Member of the Conasauga Group. The aquitard zone includes all rock formations other than those comprising the Knox Group. For this analysis, an environmental setting is included for each of these zones, designated as the Knox and Aquitard settings. Measurements of thicknesses, total porosities, fracture porosities, and matrix conductivities are discussed in Moore and Toran (1992). Specific yields were computed from hydrograph analysis for Ish Creek and the headwaters of the Melton Branch. Storage capacity was calculated from specific yield and average water-table fluctuation. Conductivity was calculated from the average transmissivity, as measured from slug tests and the thickness of each unit. Hydraulic gradient was measured on water-table maps. The hydraulic gradient and the conductivity were used to estimate groundwater velocities.

The meteorological and atmospheric dispersion data in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability used in this analysis for ORNL are presented in Bergenback et al. (1995).
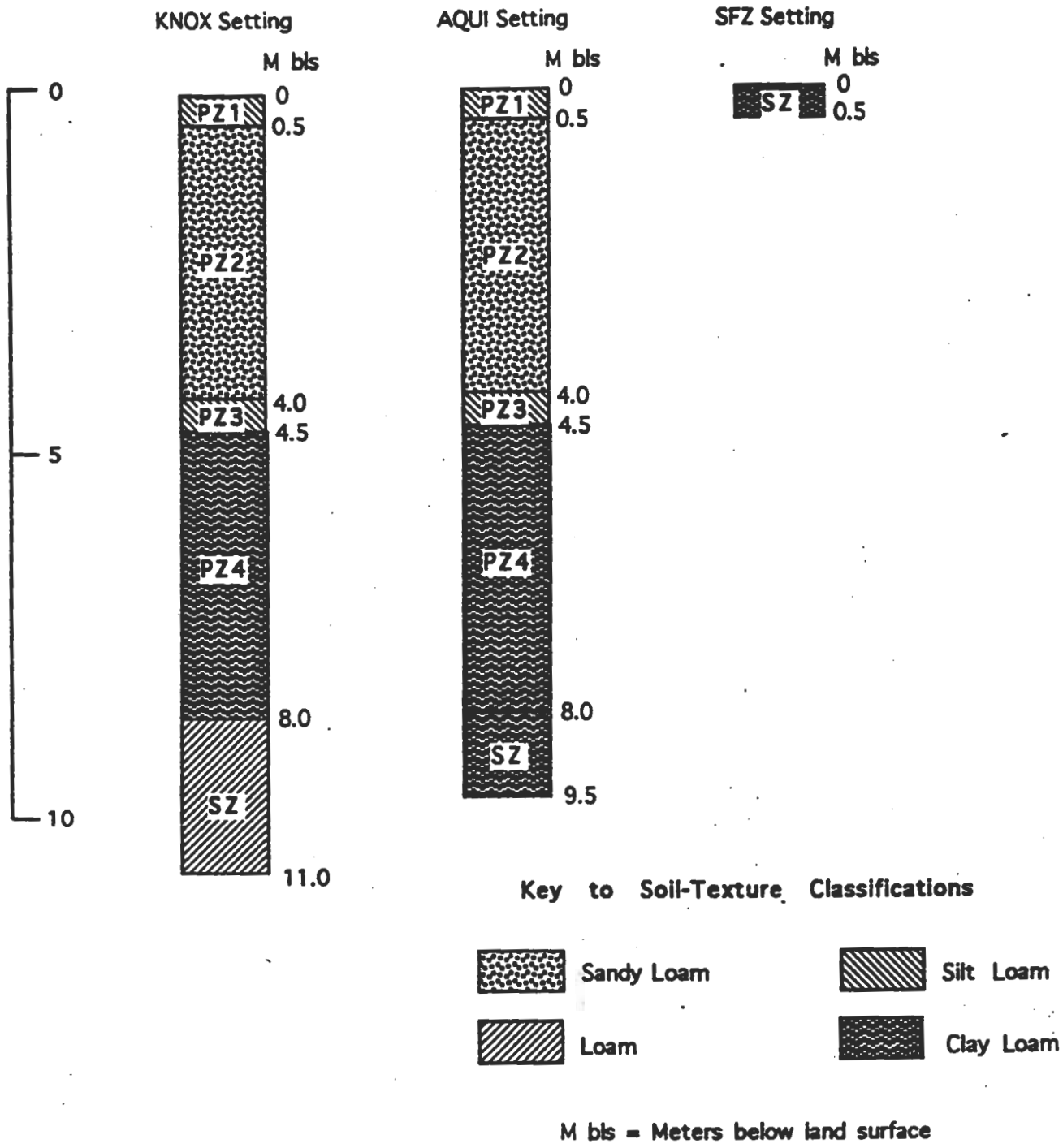
Figure C.5. Stratigraphic Columns for Oak Ridge National Laboratory Environmental Settings

C.11

C.6 Rocky Flats Environmental Technology Site

The RFETS is located on Quaternary-aged alluvial and fluvial deposits. The Rocky Flats alluvium is the dominant unit associated with the broad, flat-lying areas on the western half of the site. The thickness of the alluvium varies across the site. Near the western boundary, the alluvium can exceed 15 m but gradually pinches out toward the middle of the site. Surficial deposits in the central portions of the plant are primary colluvium. These deposits are generally thin and tend to cover the slopes and floors of ravines located north and south of the main facility. Valley fill, which is essentially fluvially reworked Rocky Flats alluvium, is found in the eastern portion of the site. Bedrock in the vicinity of the plant is of Cretaceous age and dips gently toward the east. The uppermost unit is the Arapahoe Formation, a light to medium olive-gray sequence of claystones, siltstones, and sandstones. In general, the Arapahoe Formation has a low permeability and, hence, a major influence on the hydrology in the area. Directly beneath the Arapahoe Formation is the Laramie Formation, a thick sequence consisting predominantly of claystones with a few water-bearing sand lenses. The Fox Hills sandstone (a confined aquifer) underlies the Laramie Formation.

Groundwater of concern is primarily the shallow aquifer overlying the Arapahoe Formation and is found within the Rocky Flats alluvium and valley fill. This unconfined aquifer is recharged directly by deposition in the vicinity of the RFETS and from areas west of the installation but east of the Front Range. Groundwater discharges to surfacewaters at seeps and springs along both Woman's and Walnut creeks.

One of the five hydrogeologic environmental settings used to represent the RFETS in Holdren et al. (1995). Figure C.6 presents stratigraphic columns for each of these settings.

As previously described, the RFETS is built on alluvial materials derived from the Colorado Front Range. The bedrock in the area is gently dipping Cretaceous clay and siltstones, the uppermost of which is the Arapahoe Formation. Because of the impervious nature of these units, contaminant migration has been limited to the upper area that is divided into two separate settings, one in which Rocky Flats alluvium is modeled as the surficial material and one in which the more-permeable colluvium is modeled at the surface. Because colluvium is generally found in sideslopes and floors of the ravines, the thickness of the partially saturated zones associated with the colluvium are thinner than those modeled for the Rocky Flats alluvium. The fifth environmental setting was developed for Operable Unit 3, located east of the installation and adjacent to the Great Western Reservoir and Standley Lake. This setting describes what is essentially the reworked Rocky Flats alluvium and is distinct because of the relatively shorter distances to drinking-water supplies. Surfacewater data are presented in Holdren et al. (1995, Table 23.1). Hydrogeologic data for the various settings are also presented in Holdren et al. (1995, Table 23.2 through 23.5).
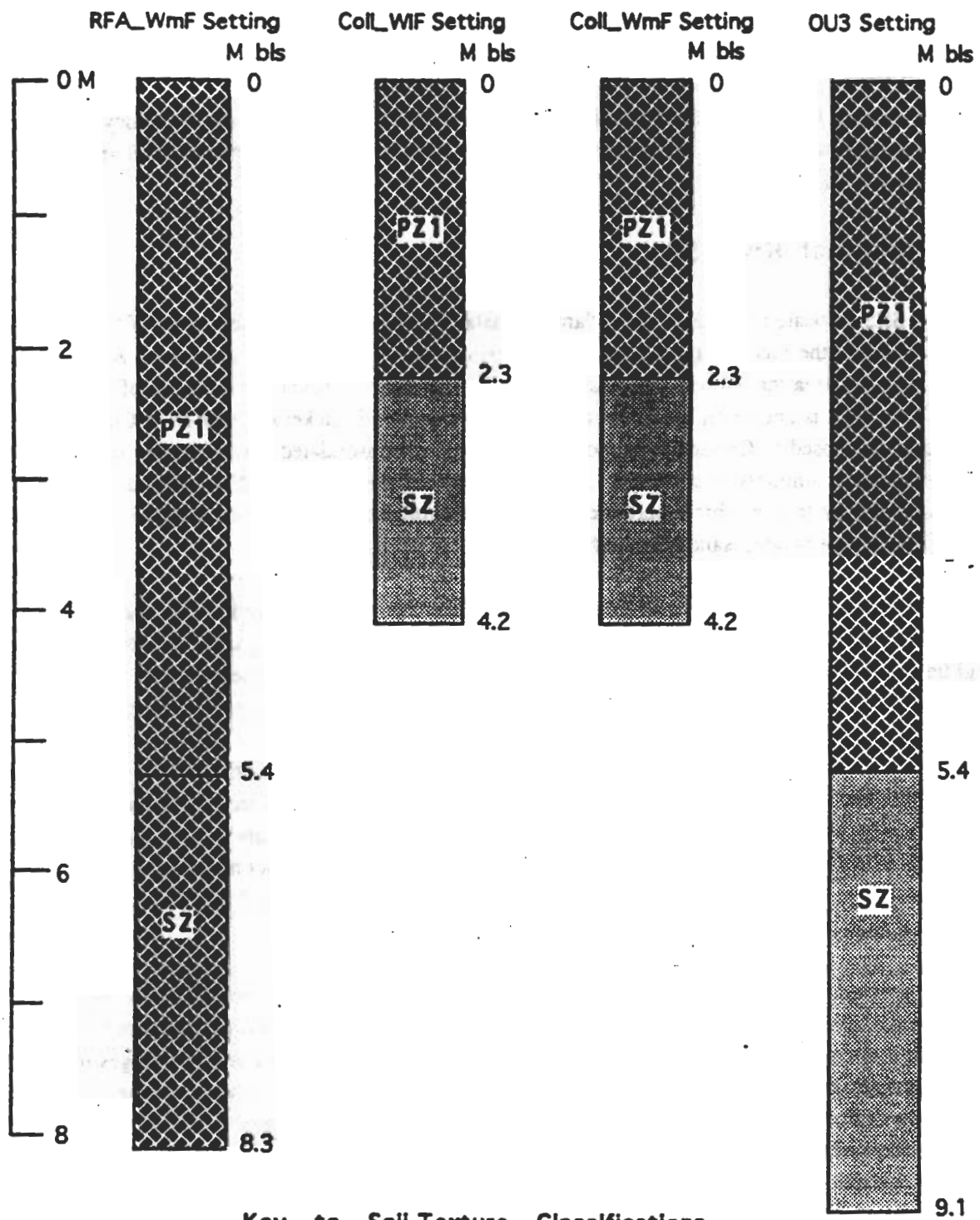
Key to Soil-Texture Classifications

Sand   Sandy Clay

M bls = Meters below land surface

Figure C.6. Stratigraphic Columns for Rocky Flats Environmental Technology
Site Environmental Settings

The meteorological and atmospheric dispersion data in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability used in this analysis for RFETS are presented in Bergenback et al. (1995).

C.7 Savannah River Site

The SRS is located on the Upper Atlantic Coastal Plain, about 32 km southeast of the Fall Line, which separates the Piedmont and Coastal Plain provinces. The site is located on the Aiken Plateau, a relatively flat area that slopes southeastward and is dissected by several tributaries of the Savannah River. The SRS is underlain by a 210- to 370-m-thick seaward-thickening wedge of Coastal Plain sediments composed of Cretaceous to post-Eocene-aged unconsolidated sands, clayey sands, sandy clays, and lesser amounts of calcareous sediment. The sediments at the SRS constitute a multilayer hydrogeologic system in which there are clay-rich confining units partially or totally isolating transmissive zones of interbedded sand and sandy clay.

Three major water-bearing zones in the Coastal Plain sediments are divided by regional aquitards. In ascending order, these are the Cretaceous zone, the lower Eocene Congaree-Fourmile zone, and the middle to post-Eocene upper saturated zone (Arnett et al. 1993). The deepest aquifer monitored at the SRS occurs in Cretaceous-aged sediments. Overlying and separating the deepest zone from the intermediate Congaree-Fourmile aquifer is a Paleocene-aged regional aquitard. The intermediate aquifer is an important local source of groundwater. A local aquitard, referred to as the "Green Clay," overlies the intermediate aquifer, separating it from the upper saturated zone. Sediments of the upper saturated zone generally yield low amounts of water, occurring predominantly in sandy clays and calcareous sediments of middle to late-Eocene and post-Eocene age. Numerous, discontinuous, clay-rich layers act as local aquitards, creating groundwater-flow patterns at the SRS, which are locally complex. Few of these aquitards are continuous across the SRS (Arnett et al. 1993).

Groundwater-flow direction at the SRS is typically controlled by the streams on the Aiken Plateau. Groundwater in the deep Cretaceous zone flows toward and discharges into the Savannah River. Groundwater in the intermediate Congaree-Fourmile aquifer flows toward and discharges into either Upper Three Runs Creek or the Savannah River, with the closer channel having the greatest influence. Groundwater in the upper saturated zone flows toward and discharges into the nearest perennial stream, swamp, or river. Local flow patterns may be very complex; nevertheless, all groundwater in these zones eventually exits the subsurface and flows from springs or streams into the Savannah River (Arnett et al. 1993).

Figure C.7 presents stratigraphic columns for each of these settings. Hydrogeologic data for the various settings are presented in Holdren et al. (1995, Tables 25.1 through 25.4). A brief description of the hydrogeology of the GS-TNX-D Area setting for SRS follows.

Setting descriptions for GS-TNX-D, K-C, and L Area settings are nearly identical to the A-M Area setting, with the exception of groundwater-flow direction and discharge within each specific area.
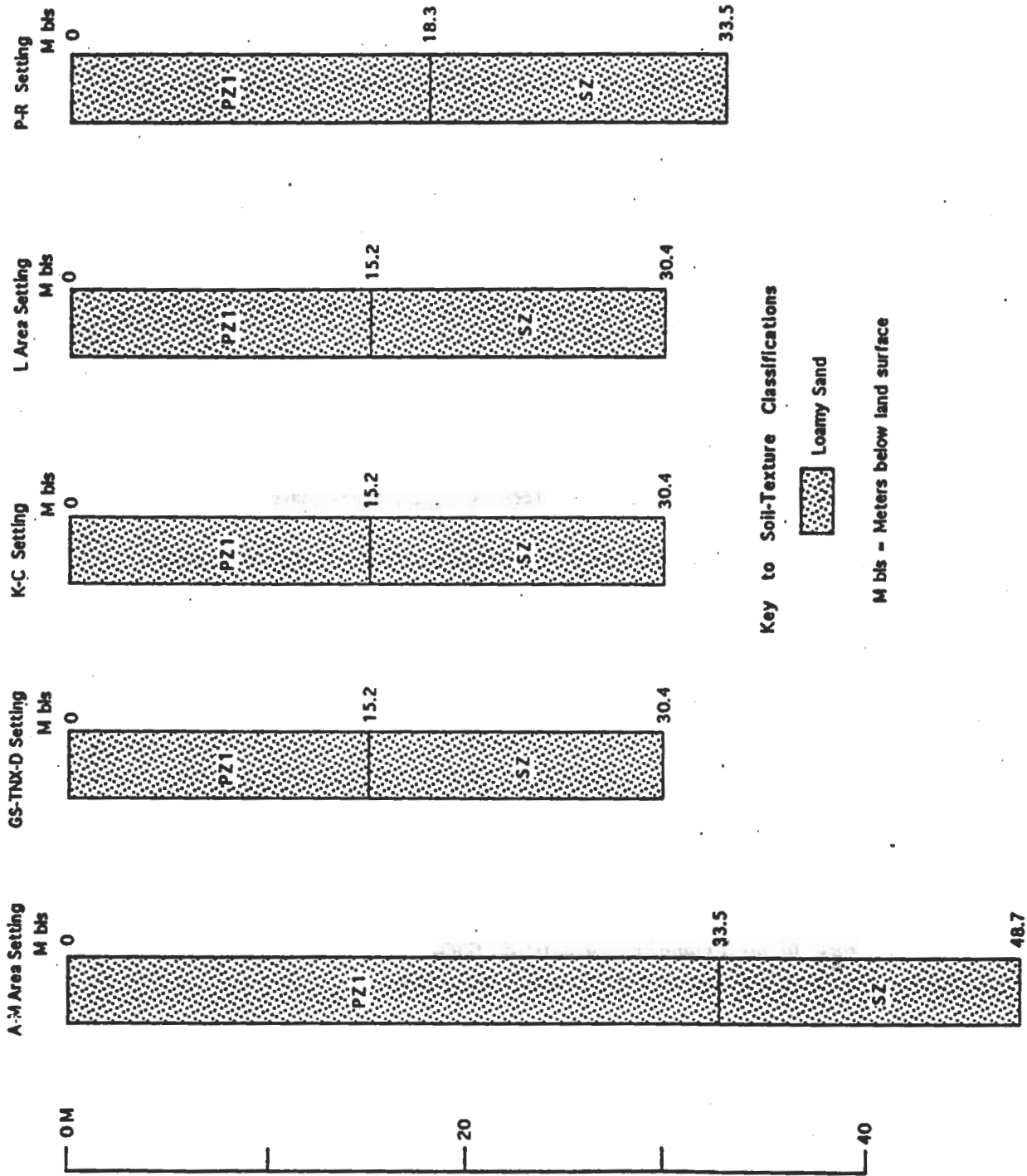
Figure C.7. Stratigraphic Columns for Savannah River Site Environmental Settings

C.15

More than a fourth of the SRS falls within the A-M Area setting, with the Pleistocene terraces occupying the southwestern quarter of the setting's area. Shallow groundwater in this setting discharges to the Upper Three Runs and Tinker creeks at springs and seeps along their margins.

The units of interest are limited to the saturated zone and uppermost aquifer. The thickness of the saturated zone averages 34 m, with maximum thicknesses reaching more than 46 m at higher elevations. The upper aquifer averages 15 m in the A-M Area, but may be as thick as 46 m. A total porosity value of 40% and an effective porosity of 25% were used for the A-M Area setting. The vertical hydraulic conductivity of the saturated zone is estimated at 1.0E-04 cm/s. Hydraulic gradient ranges from 0.016 to 0.002, based on measurements from a 1993 water-table map. The effective porosity value, hydraulic conductivity, and hydraulic gradient were used to calculate the groundwater-flow velocity, which ranged from 3.0E-05 to 3.3E-06 m/d. The remaining values for the A-M Area setting are based on default values for the soil types. This setting description will focus on items unique to the individual areas, as well as information pertaining to all three areas.

Pleistocene terraces occupy the southwestern third of the GS-TNX-D Area, and the Aiken Plateau occupies the upland two thirds of this setting. Water in the shallow aquifer flows northward from all three operation units and discharges to Upper Three Runs Creek. There is also a component of southward flow to Four Mile Creek. Discharge is at springs and seeps along the creek margins.

The meteorological and atmospheric dispersion data in the form of a joint-frequency distribution of wind direction, wind speed, and atmospheric stability used in this analysis of SRS are presented in Bergenback et al. (1995).

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Appendix D

No Action Alternatives 1A and 1B Analysis

Appendix D

No Action Alternatives 1A and 1B Analysis

The long-term impacts from No Action Alternatives 1A and 1B were estimated based on a qualitative analysis using results from the No Action Alternative 2 analysis. For No Action Alternative 1A, six generator sites were analyzed, while No Action Alternative 1B was analyzed for four generator sites. Tables D.1 and D.2 provide the inventories associated with each site for No Action Alternative 1A and No Action Alternative 1B, respectively. All the transport and exposure assumptions used in No Action Alternative 2 were used to evaluate No Action Alternatives 1A and 1B. All source assumptions were the same as No Action Alternative 2 except the inventories used to estimate the impacts and the number of sites evaluated based on the different alternative. Tables D.1 and D.2 also provide the LCFs estimated from this qualitative analysis for basic, additional and summed inventories. Inorganic and organic contaminants were not evaluated for No Action Alternatives 1A and 1B because the analysis was only interested in 10,000 year aggregate LCFs from radiation exposure.

Table D.1. Basic and Additional Inventory Volumes and Aggregate LCFs over 10,000 Years for No Action Alternative 1A

No Action Alternative 1A Inventories and 10,000-Year Aggregate LCFs						
Six Generator Sites	Basic Inventory (m³)	Additional Inventory (m³)	Total Inventory (m³)	Aggregate LCFs over 10,000 Years		
				Basic LCF	Additional LCF	Summed LCF
Hanford	37000	22920	59920	7.5E-07	4.6E-07	1.2E-06
INEEL	10000	31000	41000	6.2E-05	1.9E-04	2.6E-04
LANL	2400	4900	7300	8.7E-06	1.8E-05	2.6E-05
ORNL	1800	960	2760	1.1E-07	5.6E-08	1.6E-07
RFETS	3800	0	3800	4.9E-04	0.0E+00	4.9E-04
SRS	5000	1800	6800	3.0E-07	1.1E-07	4.1E-07

Table D.2. Basic and Additional Inventory Volumes and Aggregate LCFS over 10,000 Years for No Action Alternative 1B

No Action Alternative 1B Inventories and 10,000-Year Aggregate LCFs						
Four Generator Sites	Basic Inventory (m³)	Additional Inventory (m³)	Total Inventory (m³)	Aggregate LCFs over 10,000 Years		
				Basic LCF	Additional LCF	Summed LCF
Hanford	37000	22920	59920	7.5E-07	4.6E-07	1.2E-06
INEEL	22000	36000	58000	1.0E-04	1.7E-04	2.7E-04
ORNL	1800	960	2760	1.1E-07	5.6E-08	1.6E-07
SRS	5000	1800	6800	3.0E-07	1.1E-07	4.1E-07

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