AR TARGET SHEET

The following document was too large to scan as one unit, therefore, it has been broken down into sections.

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4.0 RESULTS OF ANALYSES

4.1 INTRODUCTION

This chapter presents the results of the analyses described in Chapter 3. It also discusses the data and methods affecting those results. This chapter emphasizes understanding the suite of analyses rather than the computational results as such. The detailed results and additional information on the calculational results can be obtained from the following documents that are combined in Puigh 2001:

- *Waste Form Release Calculations for the 2001 Immobilized Low-Activity Waste Performance Assessment* (Bacon 2001)
- *Near-Field, Far-Field, and Estimated Impact Calculations for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment: 2001 Version* (Finfrock 2000b)
- *Groundwater Flow and Transport Calculations Supporting the Immobilized Low-Activity Waste Disposal Facility Performance Assessment* (Bergeron 2000)

This chapter is divided into four parts, organized by scenario:

- General Comments (Section 4.2)
- The results from the groundwater scenarios, including the impact on surface water (Sections 4.3 - 4.11)
- The results from the releases to air (Section 4.12)
- The results from biotic pathways (Section 4.13)
- The results of catastrophic events (Section 4.14)
- The ALARA analysis (Section 4.15).

For each scenario, the contaminant concentrations in drinking water and the all-pathway dose are calculated. Chapter 7 provides comparisons with performance objectives.

4.2 COMMENTS ON CALCULATIONS

4.2.1 Presentation of Results

This subsection describes how the results will be presented for each major subset of calculations:

- Moisture flow into the disposal facility
- Contaminant release from the waste form and into the vadose zone
- Moisture flow and contamination transport in the vadose zone
- Moisture flow and contaminant transport in the groundwater
- Integration of calculations.

Because of the long times considered in this analysis, moisture flux into the disposal facility was considered time independent. Calculations in the previous ILAW performance Assessment (Mann 1998a) show that moisture content reaches equilibrium conditions relatively rapidly and transient moisture effects (during construction, for example) have relatively little impact on the results. Also, most of the cases considered have a spatially independent moisture influx into the system. Thus, moisture flow into the disposal facility is presented as a single number (i.e., the result is time and space independent). For the other cases, calculated moisture flows are displayed graphically.

Because the waste form calculations treat moisture flow, chemical reactions, waste form degradation, and contaminant release, as well as contaminant transport, the calculations were quite time consuming. Thus, most results come from one-dimensional calculations. In general, results are displayed as a contaminant flux leaving the disposal facility as a function of time, generally showing a relatively fast build up to a nearly constant value.

Because of the disposal facility's size along the "long" dimension, vadose zone calculations can be reduced to two dimensions. However, because of the vadose zone's thickness and the dispersion of contaminants as they travel to the groundwater, the contaminant flux entering the groundwater does not significantly vary spatially. Thus, most results of contaminant flux entering the groundwater will not be displayed as a function of distance from the disposal facility's centerline.

In most vadose zone calculations for other disposal sites, the time during which release occurs is short relative to the travel time in the vadose zone. This results (see Section 4.9.2) in peaked contaminant releases to the groundwater. That is, the rate at which contaminants enter the groundwater rises, reaches a peak then declines somewhat more slowly than the initial rise. However, because of the slow rate that the glass waste form degrades, the results of most of these simulations show a different temporal behavior. In these simulations, the time dependence is determined by the rate of waste form degradation rather than vadose zone travel time. Thus, the rate at which contaminants enter the groundwater looks more like a step function. The time of the step is determined by the travel time in the vadose zone and by the time needed for moisture to start attacking the waste packages, which in this analysis is assumed to be zero. The shape of the

rise is determined mostly by vadose zone properties, but is relatively unimportant when addressing performance objectives, since such objectives address peak values. The height of the step is determined by the contaminant release rate from the disposal system because the vadose zone does not store mobile or near-mobile contaminants; it only delays their reaching the groundwater.

Given that the contaminants enter the groundwater at a fairly constant rate (after a delay, due to travel in the vadose zone) and that the travel times in the groundwater are small compared to those in the vadose zone, the groundwater calculations use a time-independent source to calculate groundwater effects. Although some plume maps are provided, most groundwater results are presented as well intercept factors which are defined as the ratio of the contaminant concentration at the point of calculation (expressed as Ci/m^3) to the contaminant concentration entering the groundwater (expressed as Ci/m^3). The key parameters driving the well intercept factor are the groundwater flow velocities and the distance from the disposal facility. Groundwater flow velocities are themselves determined by geology, hydraulic conductivities, and differences in hydraulic "pressure" (including pumping speed at the well location).

The integration of calculations includes the adding of inventory and dosimetry information. The temporal shape of the results is dominated by the waste form release calculations. Thus, although time- and space-dependent information is displayed, for most of the sensitivity cases, the calculated impacts are tabulated only at 1,000 years, 10,000 years, and the time of maximum value.

All positions on the Hanford Site depicted in these sections are described using the Hanford coordinate system. One unit in this system equals 1 meter. Thus, distances are easily inferred.

4.2.2 Calculational Accuracy and Stability

All of the calculations were very stable and did not suffer significant mass loss. This is in contrast with the 1998 ILW PA where fractional mass balance errors of 3.5 percent occur after 50,000 years.

4.3 RESULTS OF GROUNDWATER SCENARIOS, BASE ANALYSIS AND BEST ESTIMATE CASES

The groundwater scenario is covered in many sections, so various key points can be emphasized in context.

4.3.1 Overview

This section presents the results of the analyses for the base analysis and best-estimate groundwater contamination cases. Sections 4.3.2 through 4.3.6 provide the results for the base analysis case and Sections 4.3.7 through 4.3.11 provide the results for the best-estimate case. The following sections provide the sensitivity analyses:

- Section 4.4 (moisture flow into the disposal facility)
- Section 4.5 (waste form release)

- Section 4.6 (vadose zone moisture flow and contaminant transport)
- Section 4.7 (groundwater flow and contaminant transport)
- Section 4.8 (other factors)
- Section 4.9 (extreme cases).

Section 4.10 describes the effect of other Hanford Site activities on the groundwater affected by this disposal action. Section 4.11 puts into context the information given in Sections 4.3 through 4.10.

4.3.2 Base Analysis Case: Moisture Flow Into Disposal Facility

> The surface barrier is assumed to have degraded to natural conditions; the side slope barrier remains in place; and no capillary barrier is present. Thus, the infiltration rate is 4.2 mm/year, except directly under the side slope where it increases to 50 mm/year.

The base analysis case assumes that the surface barrier has degraded to natural conditions and that no capillary barrier is present under the surface barrier. The side slope barrier connecting the surface barrier and the undisturbed soil remains, yielding a high recharge rate of 50 mm/year. Elsewhere a recharge rate of 4.2 mm/year (corresponding to Burbank loamy sands) is used. Because of the large size of the disposal facility, the small size of the side slope, and the strong vertical flow in such conditions, the moisture flow recharge rate into the facility is basically 4.2 mm/year everywhere. The recharge rate for the Burbank sandy loams is used, because it is higher than the natural recharge rate for the Rupert sands (0.9 mm/year), the other type of soil on the disposal site.

This simple case was chosen for the base analysis case not only to speed calculations, but because we recognize that the sand-gravel capillary barrier used in the best estimate case (Section 4.3.7) has not yet been designed. Because the sand-gravel capillary barrier results in much lower impacts, the base analysis case is conservative relative to the best estimate case.

4.3.3 Base Analysis Case: Waste Form Release

> The release rate from the disposal facility is based on deterministic simulation of the degradation of LAWABP1 glass and the resulting contaminant transport. The result is a small release that builds over long times, reaching 0.71 ppm/year at 10,000 years.

Time-dependent, one-dimensional calculations of waste form release and resulting contaminant transport were performed using LAWABP1 glass as the waste form. These calculations model the changing chemical environment as the waste form degrades and the various chemical reactions proceed. Figure 4-1 displays the volumetric water content (i.e., the fraction of the available pore volume occupied water) for the base case as a function of vertical position in the disposal facility. Because of the high sodium content of the glass, the pH inside the facility is a strong function of time and space. Figure 4-2 shows the variation of pH with vertical position for various times. Glass dissolution is a strong function of the pH in the surrounding fluid (see Section 3.4.4). Sensitivity cases are described in Section 4.5. The two-dimensional simulation is described in Section 4.5.8.

Figure 4-1. Moisture Content Inside the Disposal Facility for Various Recharge Rates. The base analysis case uses 4.2 mm/year.

The top of the disposal facility (below cap) is at 15 meters with the bottom at 4.5 m.

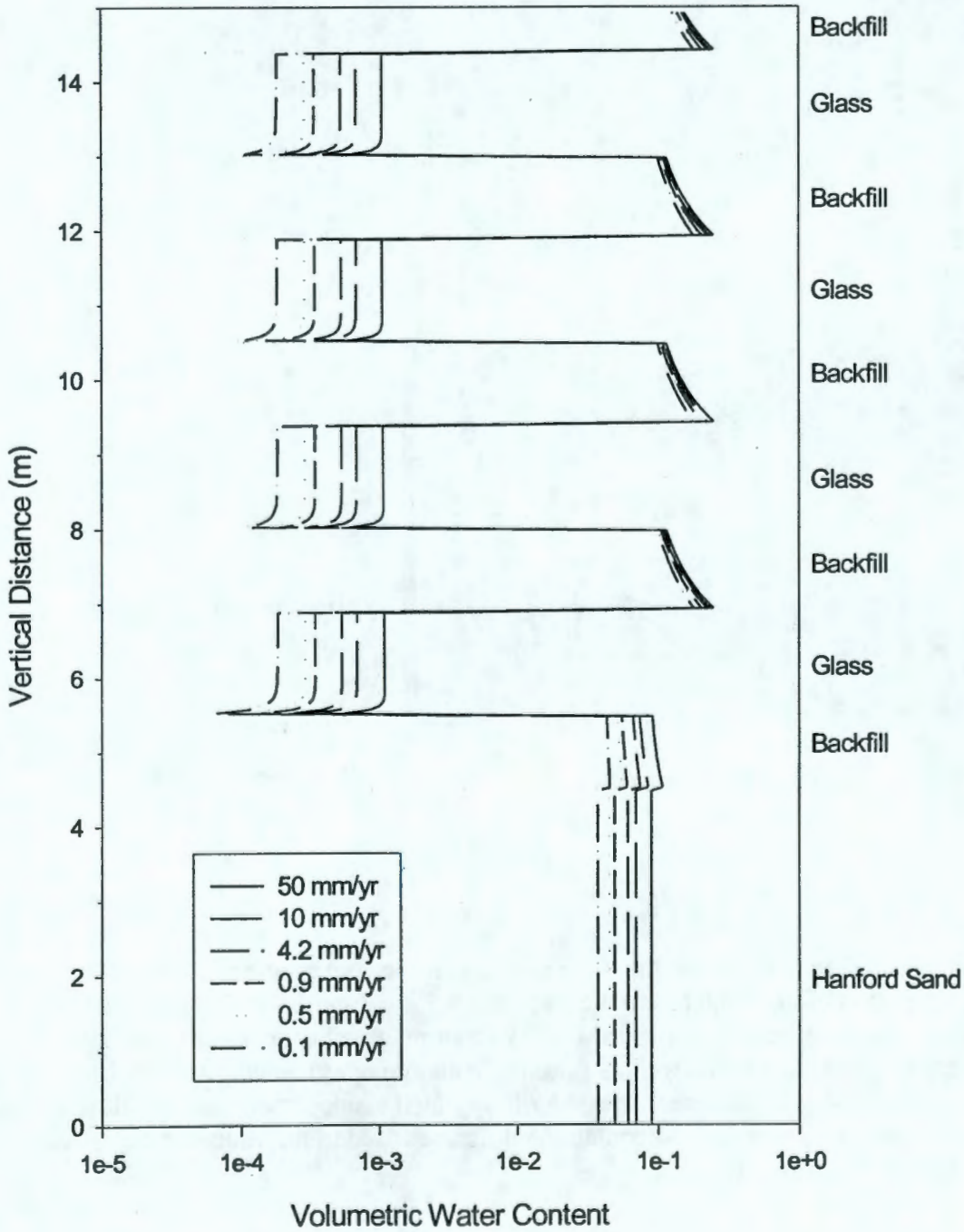


Figure 4-2. pH inside the Disposal Facility for Base Analysis Case.

The top of the disposal facility (below cap) is at 15 meters with the bottom at 4.5 m.

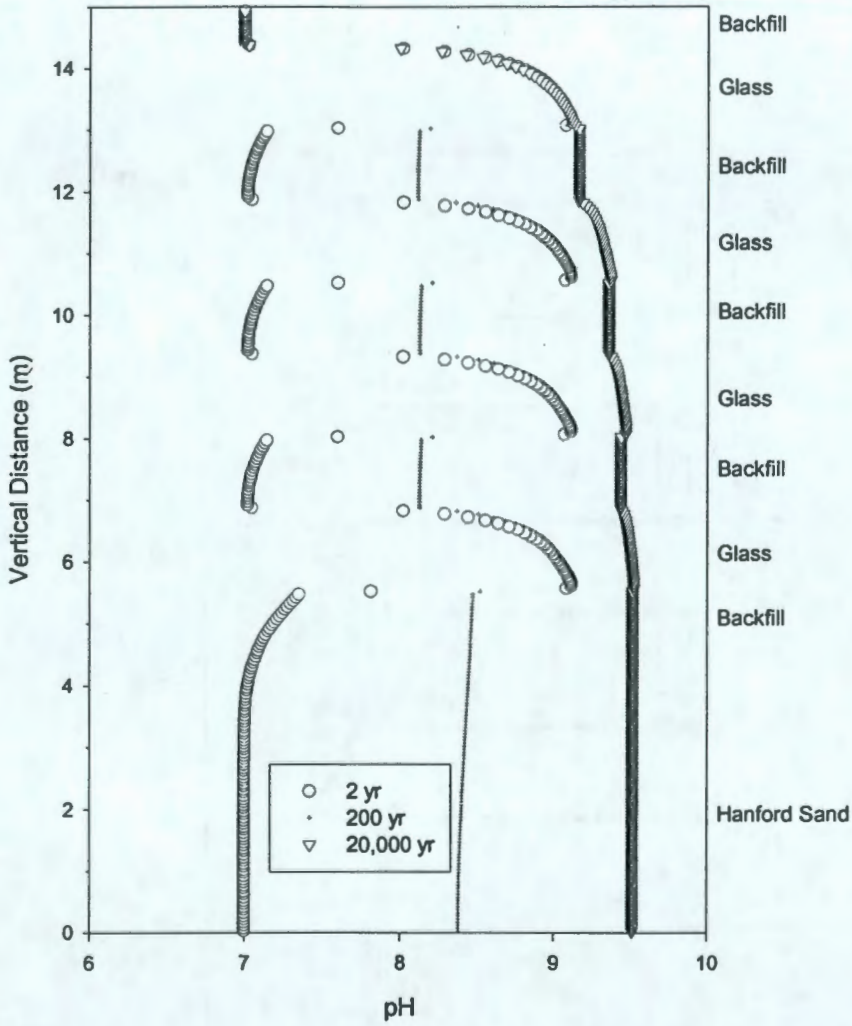


Figure 4-3 displays the results of the one-dimensional simulation of the waste form release as a function of time and space the disposal facility. Figure 4-4 displays the contaminant flux to the vadose zone from the disposal facility (assuming unit concentration initially for each contaminant) for the base case as well as for other recharge rates (ranging from 0.1 to 50 mm/year). (Section 4.5.4 discusses the effect of varying the amount of moisture flow.) Table 4-1 presents the inventory-normalized contaminant flux released to the vadose zone for various times for the base analysis case.

Figure 4-3. Waste Form Dissolution as a Function of Time and Space Inside the Disposal Facility.

The top of the disposal facility (below cap) is at 15 meters with the bottom at 4.5 m.

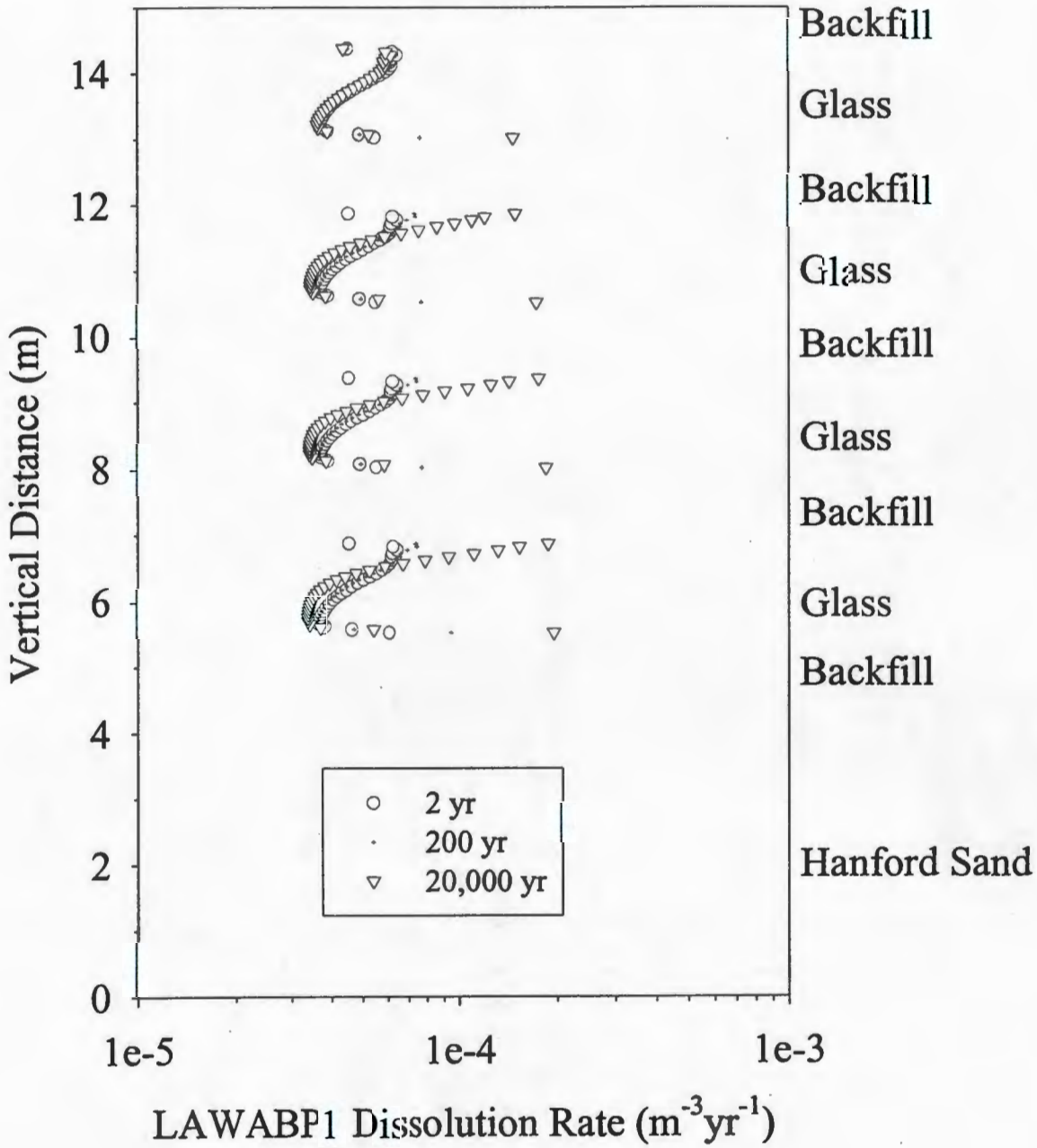


Figure 4-4. Relative Contaminant Flux to the Vadose Zone from the Disposal Facility (Assuming Unit Concentration Initially) for the Base Analysis Case and Other Infiltration Rates.

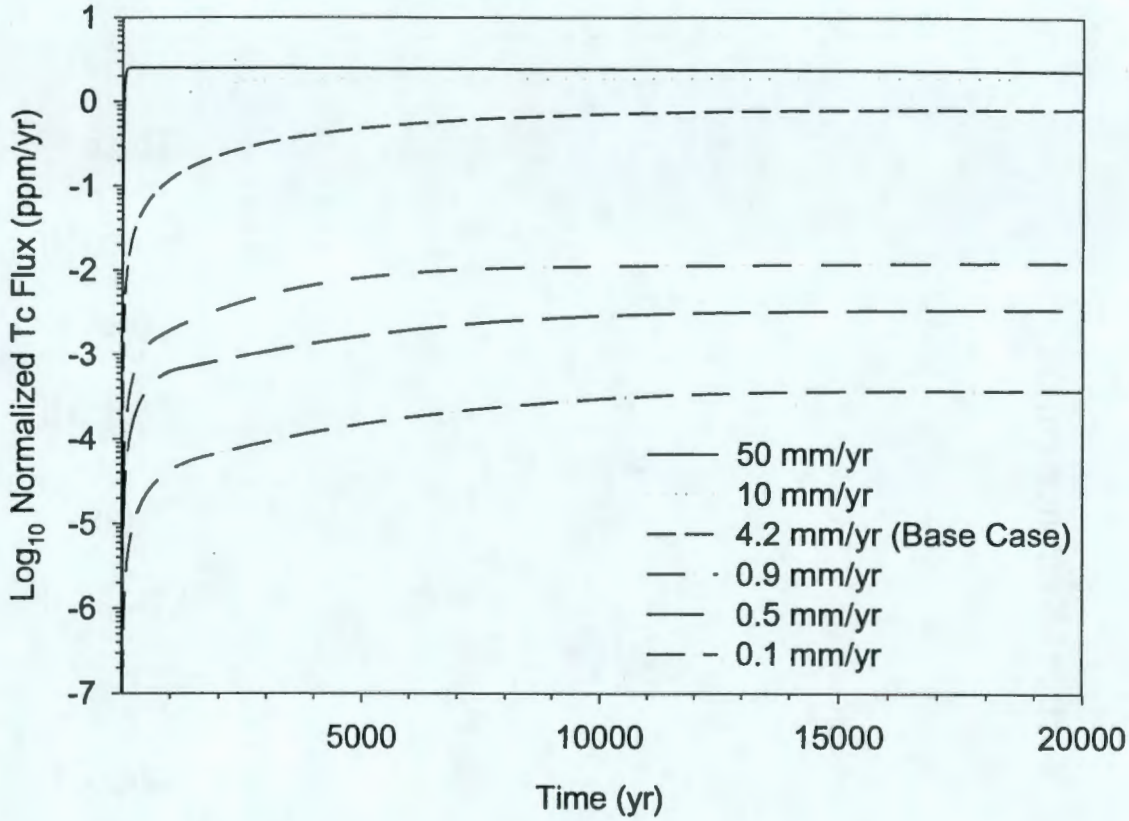


Table 4-1. Inventory-Normalized Contaminant Release to the Vadose Zone for Various Times for the Base Analysis Case.

Time (years)	Inventory-Normalized Contaminant Flux to the Vadose Zone (ppm/year)
500	0.06
1,000	0.12
2,000	0.23
5,000	0.48
10,000	0.71
20,000	0.81
50,000	0.87
100,000	0.93

The technetium flux released to the vadose zone is proportional to the TcO_4^- concentration at the lower boundary and the water flux rate. At early times, the TcO_4^- concentrations increase sharply in the glass surface layers. Glass dissolution and low water contents in the glass surface layers (Figure 4-1), coupled with a low water flux rate, causes TcO_4^- concentrations to increase rapidly in these layers. In contrast, mass transport from the glass layers is required to buildup technetium concentrations in the backfill layers. Therefore, concentrations in the backfill layers increase slowly as products of glass dissolution move from the glass layers into the backfill layers. Dilution also occurs in the backfill layers because of their much higher water content than the glass layers. Predicted glass dissolution rates (Figure 4-3) increase with time in each glass layer, but are relatively similar for each layer.

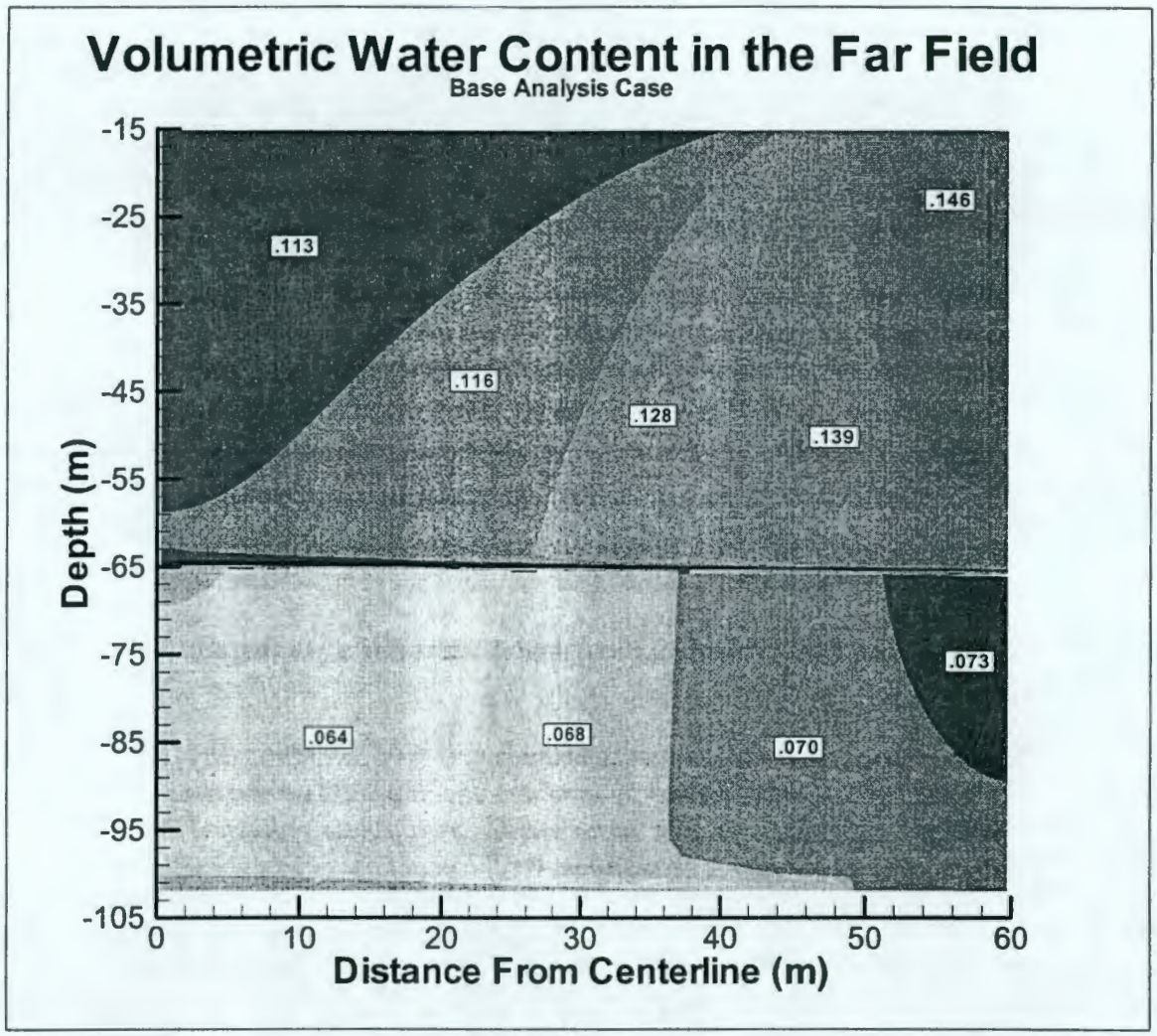
The glass dissolution rate for these simulations is highest on the edges of the glass layers. This is where the pH of the pore water is highest (Figure 4-2) and the SiO_2 (aq) concentrations are lowest. Because the glass dissolution rate is relatively low, the surface area of the glass does not decrease significantly by 20,000 years. The pH and TcO_4^- concentrations increase more rapidly in the glass layers early in the simulation, although by 20,000 years, concentrations throughout the profile are relatively similar. This indicates that at early times, the TcO_4^- flux across the lower boundary is limited by the movement rate of TcO_4^- from the glass layers.

4.3.4 Base Analysis Case: Moisture Flow And Contaminant Transport in the Vadose Zone

For the base analysis case, a time-independent surface recharge rate of 4.2 mm/year was assumed across the surface barrier, then 50 mm/year was assumed for the side slope and natural conditions (that is, for distances greater than 50 m from the trench center). The estimated moisture content for the far-field zone is shown for the base analysis case in Figure 4-5. The volumetric water content varies from 11 percent (directly beneath the disposal facility) to 15 percent (directly beneath the side slope area with a 50 mm/year recharge rate) for the Hanford formation sandy sequence, upper part of Figure 4-5. In the Hanford formation gravels, the volumetric water content is less and varies less (6 percent to 7.4 percent), lower part of Figure 4-5.

Figure 4-5. The Estimated Moisture Content for the Far-Field Zone for the Base Analysis Case.

The bottom of the disposal facility is at –10.5 m, i.e. just above the top of the figure. Distance is measured along the short dimension of the trench (See Figure 2-24).



To calculate contaminant transport, the contaminant flux leaving the disposal facility and entering the vadose zone are needed along with the moisture field. To translate the waste form results from Section 4.3.3, the moisture flux (or Darcy flow) was calculated. As seen in Figure 4-6, for the base analysis case, the moisture flux is quite constant as a function of distance from the center of the trench to the edge, where the higher recharge rates exist. Also, as seen in Figure 4-6, the capillary break (best estimate case), the capillary break significantly reduces the moisture flow through the region containing the waste packages (x dimension is less than 32m, y dimension is between -1 and -10 m). A local 1-m subsidence in the failed capillary break leads to moisture flows approaching 2 mm/year below the subsidence. Finally, a shortened capillary break (short surface barrier) leads to higher Darcy velocities in the waste package region when compared to the best analysis case. See Section 4.4 for a discussion of these sensitivity cases.

Figure 4-6. Darcy Velocity at the Top of the Far Field (Bottom of the Disposal Facility).

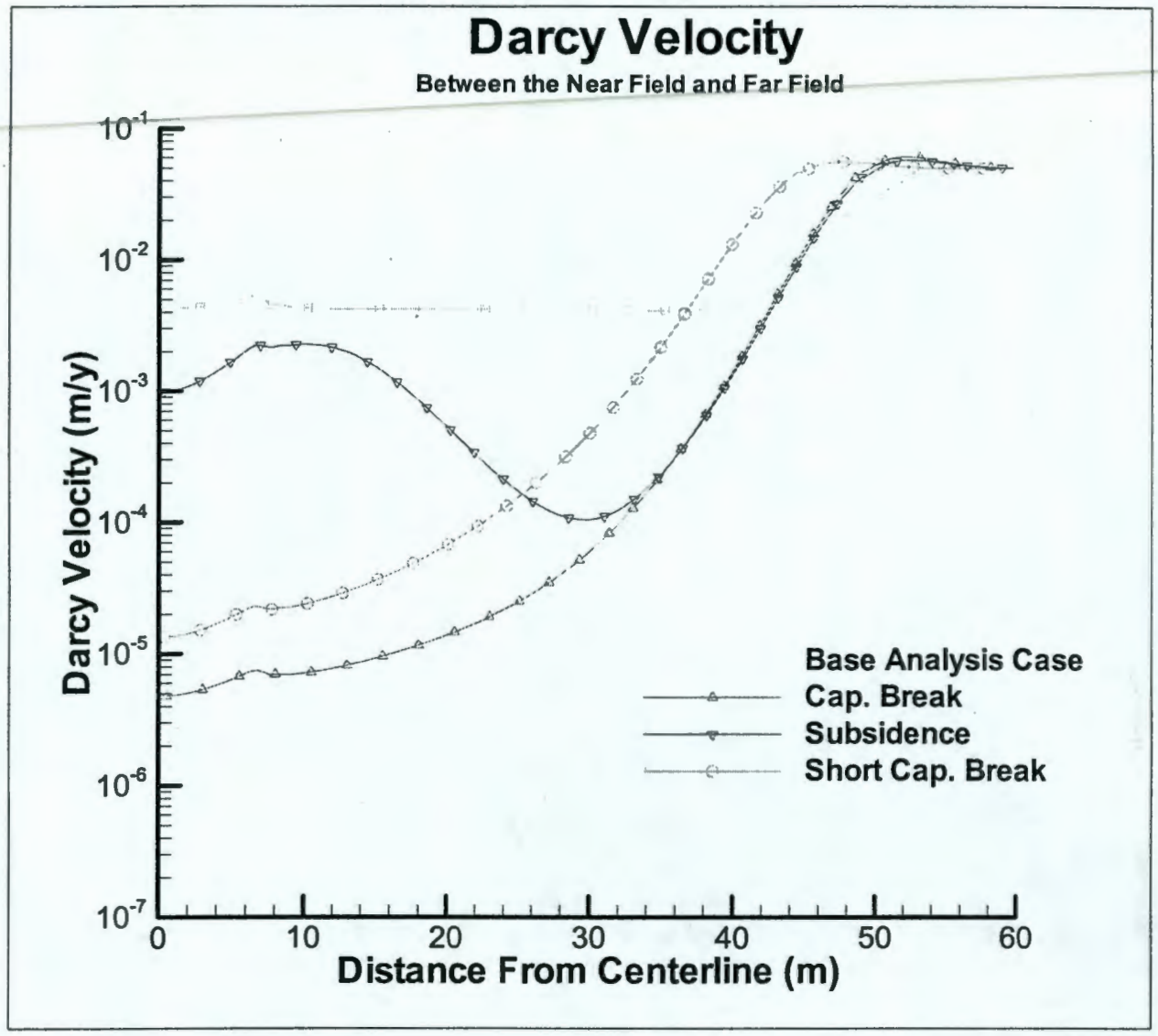
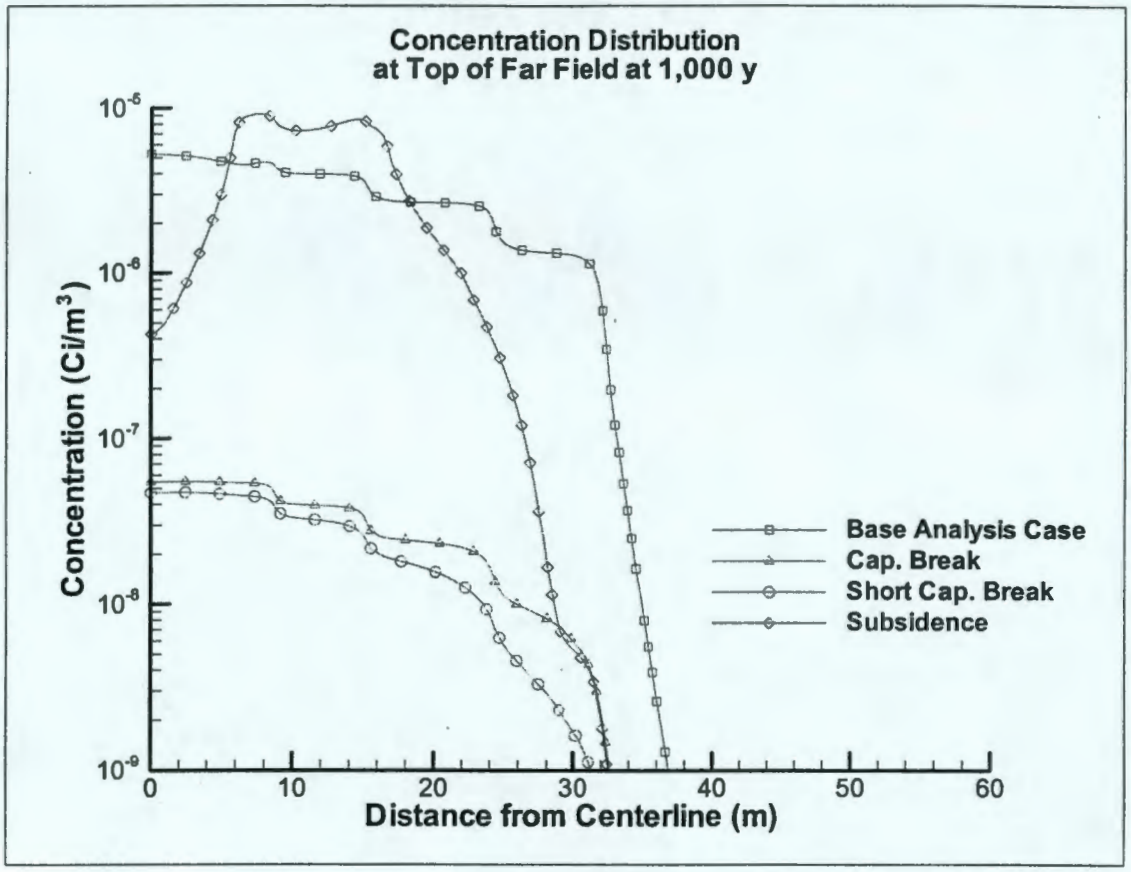


Figure 4-7 shows the result of combining the waste form release results and the moisture flow calculations. It also shows the inventory-normalized contaminant flux to the vadose zone (i.e., the flux leaving the disposal facility) at 1,000 years for the mobile contaminants for the base analysis case, the best estimate case, the shortened capillary break, and the capillary break subsidence case. The step-wise decrease in contaminant flux for the base analysis case is caused directly by the arrangement of four layers of waste packages in the disposal facility. Near the edge of the facility, fewer layers are present, so less inventory is available for release to the environment. The contaminant concentrations for the two cases where the capillary break is functioning are significantly lower than the base analysis case at 1,000 years. (For these two cases, we have assumed that the waste form release rate was equal to the waste form release rate calculated for a recharge rate of 0.1 mm/year. This assumption is conservative when calculating the contaminant transport to the aquifer. However, this assumption leads to a higher

Figure 4-7. Inventory-Normalized Mobile Contaminant Concentration at the Bottom of the Disposal Facility (i.e., Entering the Vadose Zone) at 1,000 years.



concentration at the bottom of the disposal facility for the best estimate case when compared to the shortened barrier because the contaminant release rate is the same and the shortened capillary break has a higher Darcy flux.) Finally, the case where subsidence is assumed to occur in the capillary break leads to contaminant concentrations higher than those associated with the base analysis case for a portion of the area directly beneath the discontinuity. For these sensitivity cases, the contaminant fluxes are nearer the centerline of the disposal facility than for the base analysis case because of the increased moisture flux

The contaminant flux then is calculated as a function of its position in the vadose zone, time, and K_d bin. Figure 4-8 displays the mobile contaminant concentration entering the groundwater at 10,000 years as a function of distance horizontally from the trench center. By the time the contaminants reach the groundwater, the sharp step features have dissolved. As seen in Figure 4-9, the total inventory-normalized contaminant flux entering the groundwater is similar to that leaving the disposal facility, except for the time delay.

Figure 4-8. Mobile Contaminant Concentration Just Prior to Entering the Groundwater at 10,000 Years.

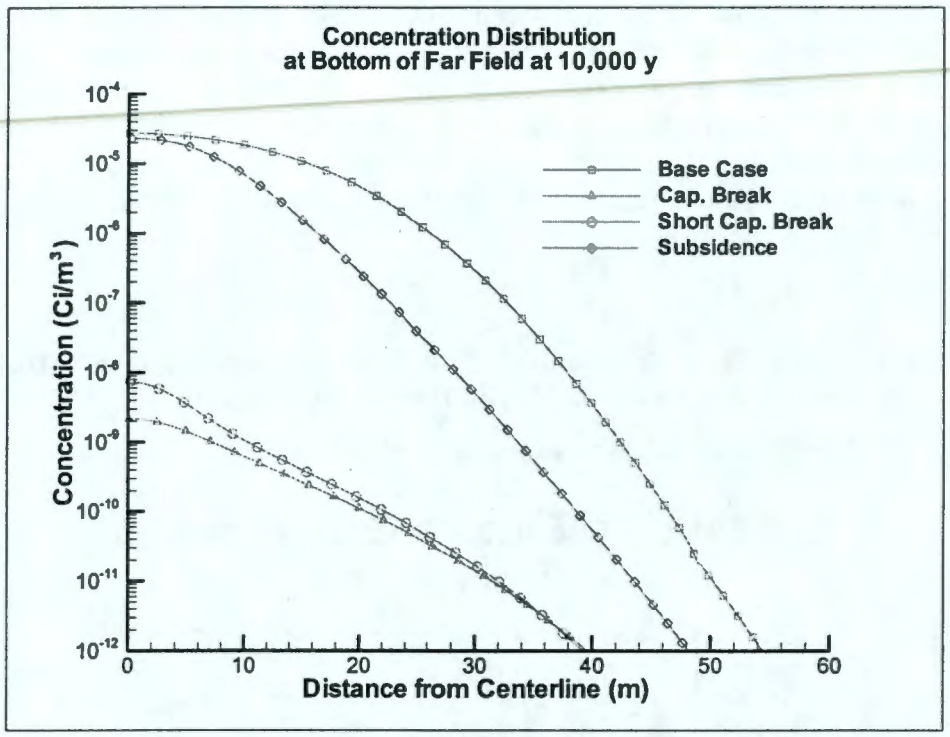


Figure 4-9. Inventory-Normalized Contaminant Flux Summed Over Horizontal Distance as a Function of Time and K_d Bin (Linear Scale).

"Facility Release" refers to values just below the disposal facility.

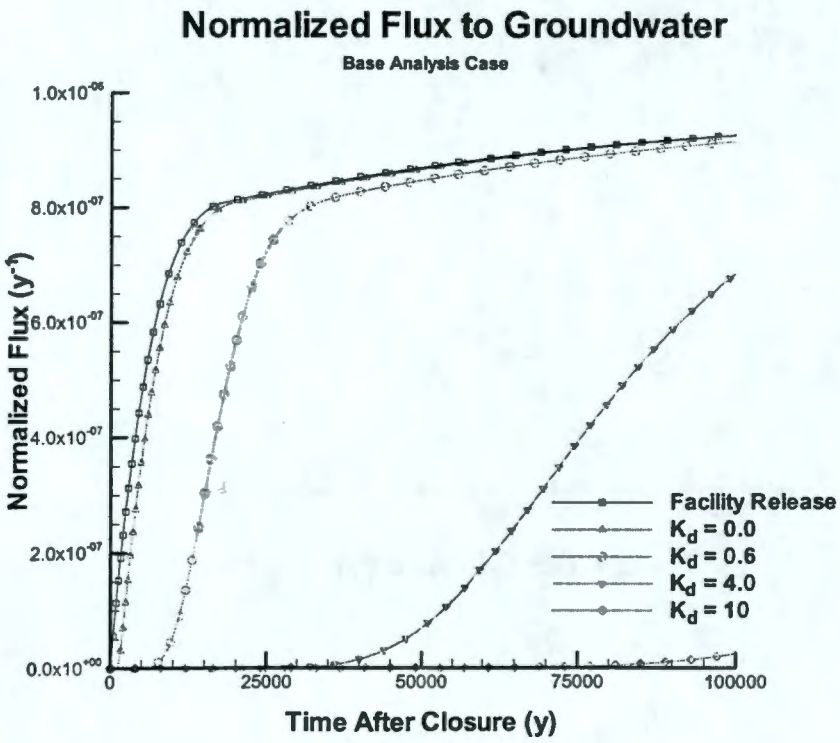
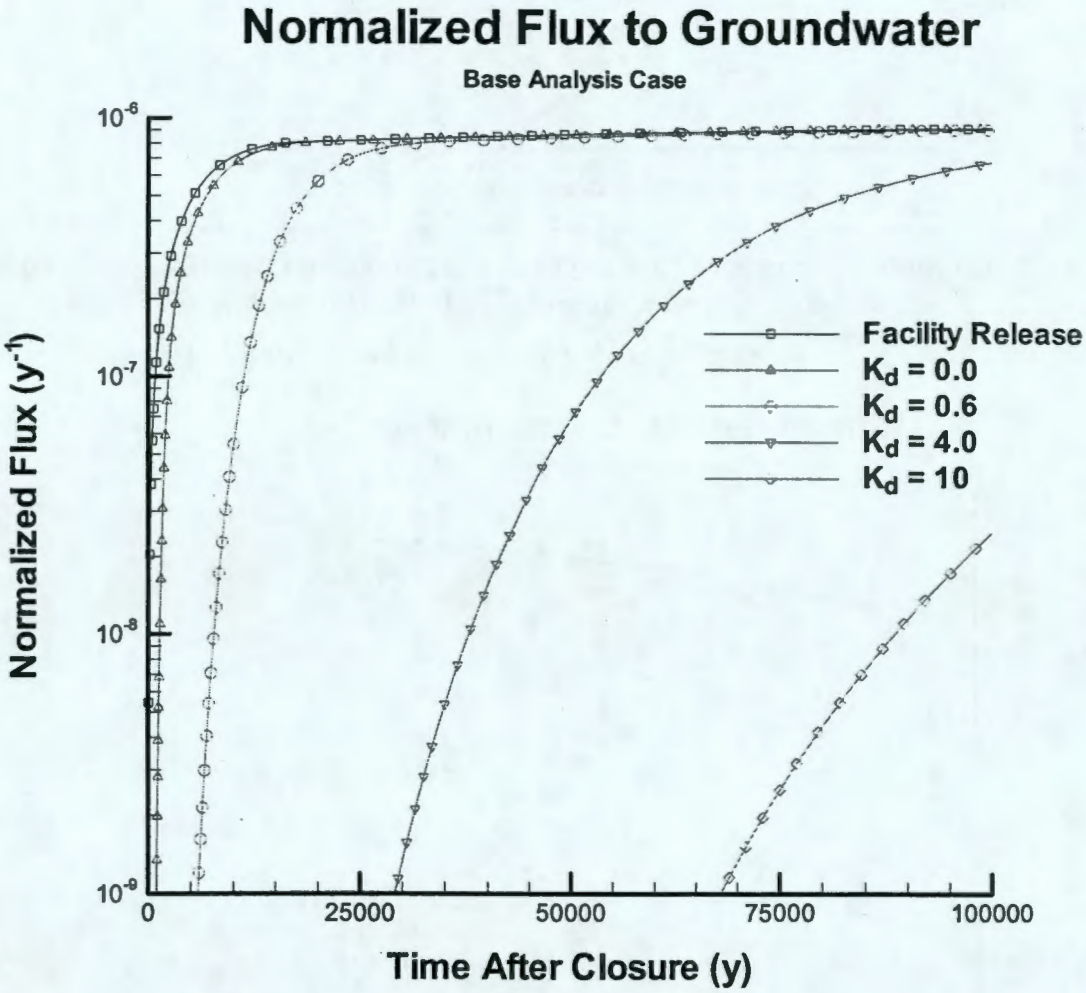


Figure 4-10 shows the contaminant flux summed over horizontal distance as a function of time and K_d bin. Only the mobile contaminants reach the groundwater during the time of compliance (the first 1,000 years). At 10,000 years, the slightly retarded contaminants ($K_d = 0.6$ mL/g) also are beginning to reach the groundwater, but their inventory-normalized contribution is still approximately 1 order of magnitude less than the mobile contribution. Higher K_d contaminants ($K_d \geq 4$ mL/g) do not contribute to the estimated doses at 10,000 years and are even less important.

Figure 4-10. Inventory-Normalized Contaminant Flux Summed Over Horizontal Distance as a Function of Time and K_d Bin (Logarithmic Scale).

"Facility Release" refers to values just below the disposal facility.

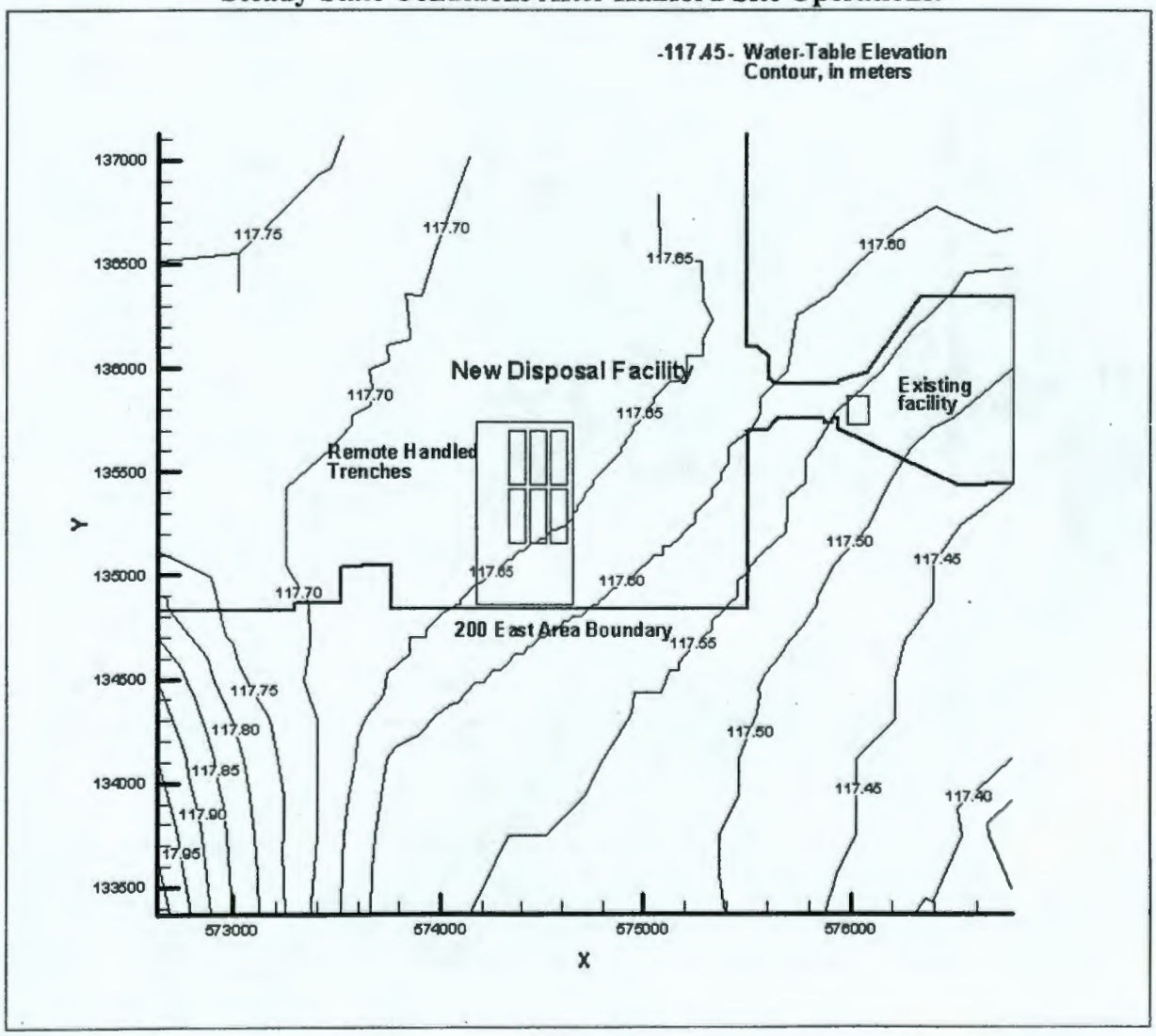


4.3.5 Base Analysis Case: Moisture Flow And Contaminant Transport in Groundwater

As noted in Section 2.8, the disposal site lies on top of the ancient channel of the Columbia River. This channel has high hydraulic conductivity and hence is a conduit for groundwater flow. Much greater dilution is estimated in this PA than in the 1998 PA (Mann 1998a).

The contaminant travel time in the vadose zone for even the most mobile contaminants will be about a thousand years. Thus, by the time the contaminants reach the groundwater, the unconfined aquifer will not be influenced by Hanford Site operations. The full Hanford Site groundwater model was used to establish boundary conditions for the local model. Figure 4-11 displays the water table elevation predicted by the local groundwater model for such future conditions.

Figure 4-11. Water Table Elevation Predicted by the Hanford Site Groundwater Model for Steady-State Conditions After Hanford Site Operations.



Using this flow field, contaminant groundwater concentrations were calculated assuming a unit contaminant flux entering the groundwater directly beneath the disposal facility. Figure 4-12 displays the resulting contaminant plume in the horizontal dimensions, while Figure 4-13 displays the vertical distribution along the line A-A' (defined in Figure 4-12), which is near the centerline of the plume.

Figure 4-12. Base Analysis Case Contaminant Groundwater Plume at the Water Table, Showing the Contaminant Concentration Values.

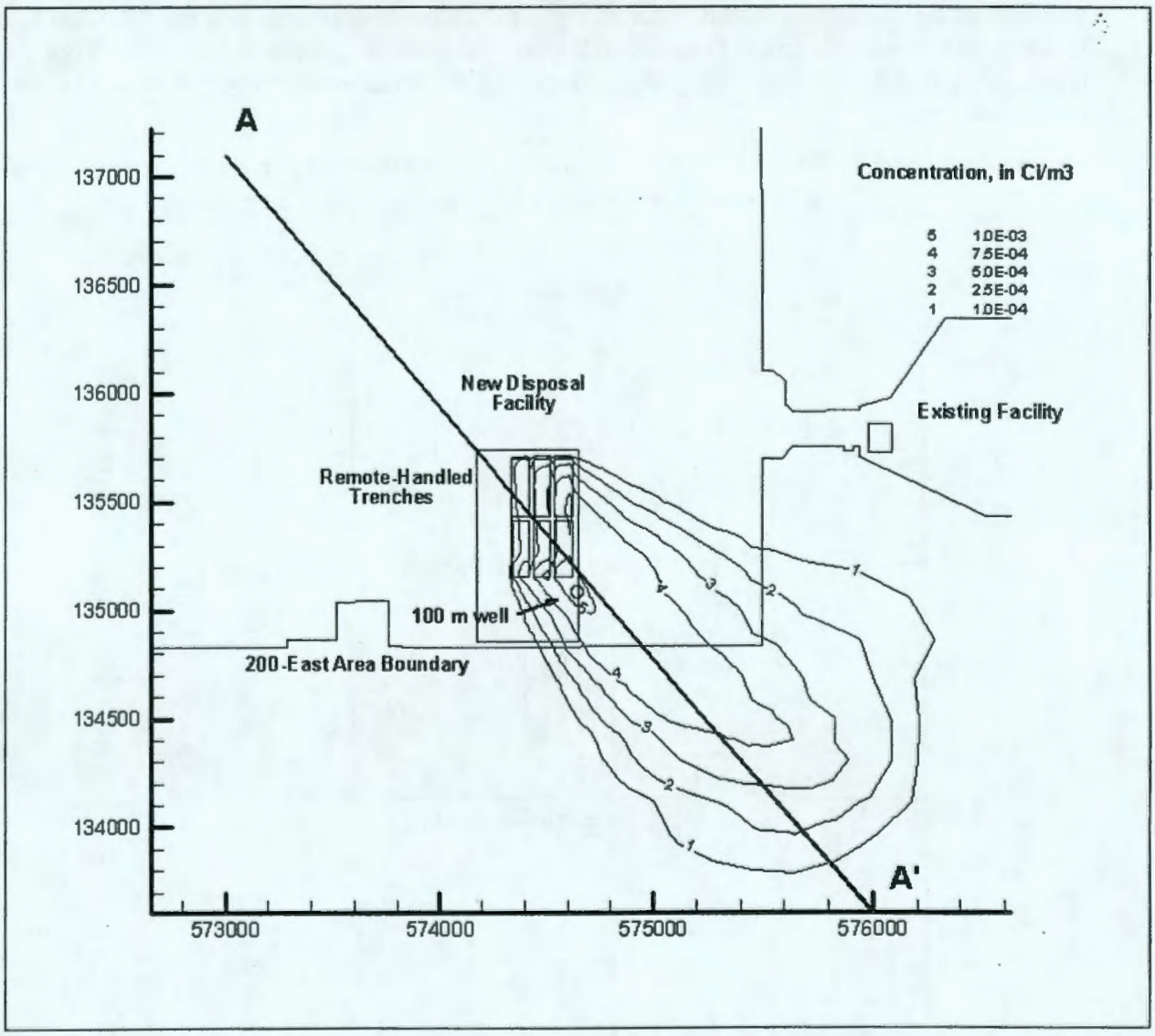
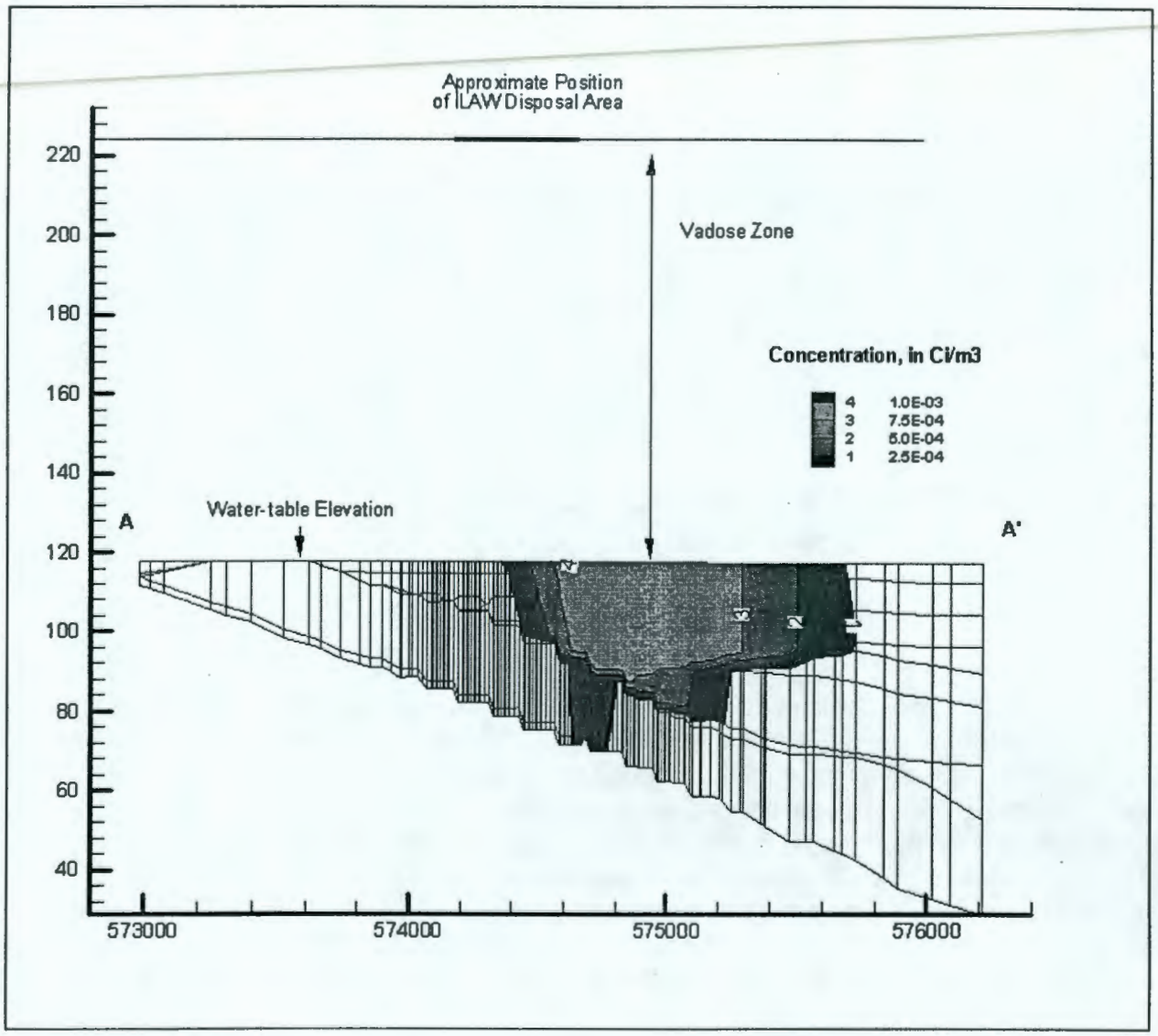


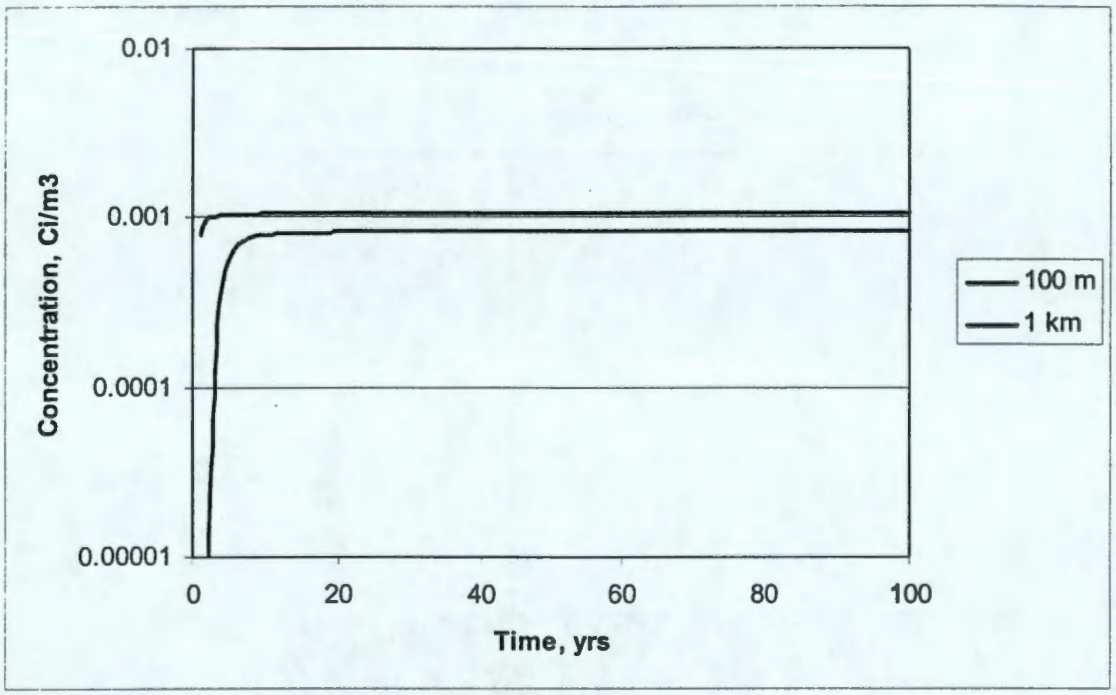
Figure 4-13. Vertical Distribution of the Groundwater Plume Along the Line A-A', Which Is Near the Centerline of the Plume.



From these results, the well intercept factor (WIF) can be calculated. The well intercept factors for wells at 100 m and 1,000 m as a function of time are displayed in Figure 4-14. The well intercept factor quickly reaches asymptotic values of 1.05×10^{-3} and 0.78×10^{-3} (respectively for the 100 m and 1,000 m wells) for trenches located in the northern part of the ILAW disposal site. That the groundwater concentration quickly reaches the asymptotic value is not unexpected given the high conductivity in the ancestral Columbia River channel and the nearness of the observation points.

The small decrease in the WIF between 100 and 1000 m is expected given the relatively short distance of transport within the underlying high permeabilities of the Hanford gravels. The effect of the high permeabilities is quite evident in Figure 4-12 and will become more evident when a different location is discussed in Section 4.7.2.

Figure 4-14. Well Intercept Factor for Wells at 100 m and 1,000 m as a Function of Time for the Base Analysis Case.



To calculate groundwater concentrations at greater distances from the disposal facility, the full-scale Hanford Site groundwater model was used. Figure 4-15 displays the asymptotic areal distribution of the contaminant plume for greater distances, while Table 4-2 shows the values. Because the full-scale model uses different grid sizes than the local scale model and because the effect of the ancestral Columbia River channel is better represented in the local scale model, the calculated groundwater concentrations differ at 1000m (5.4×10^{-4} versus 7.8×10^{-4}). Figure 4-16 provides the time history of the groundwater concentrations at the greater distances. Again the travel times (as compared to the vadose zone travel times or the waste form degradation times) are quite short. The travel time to the Columbia River is on the order of 200 to 300 years.

Figure 4-15. Asymptotic Areal Distribution of the Contaminant Groundwater Plume for Large Distances.

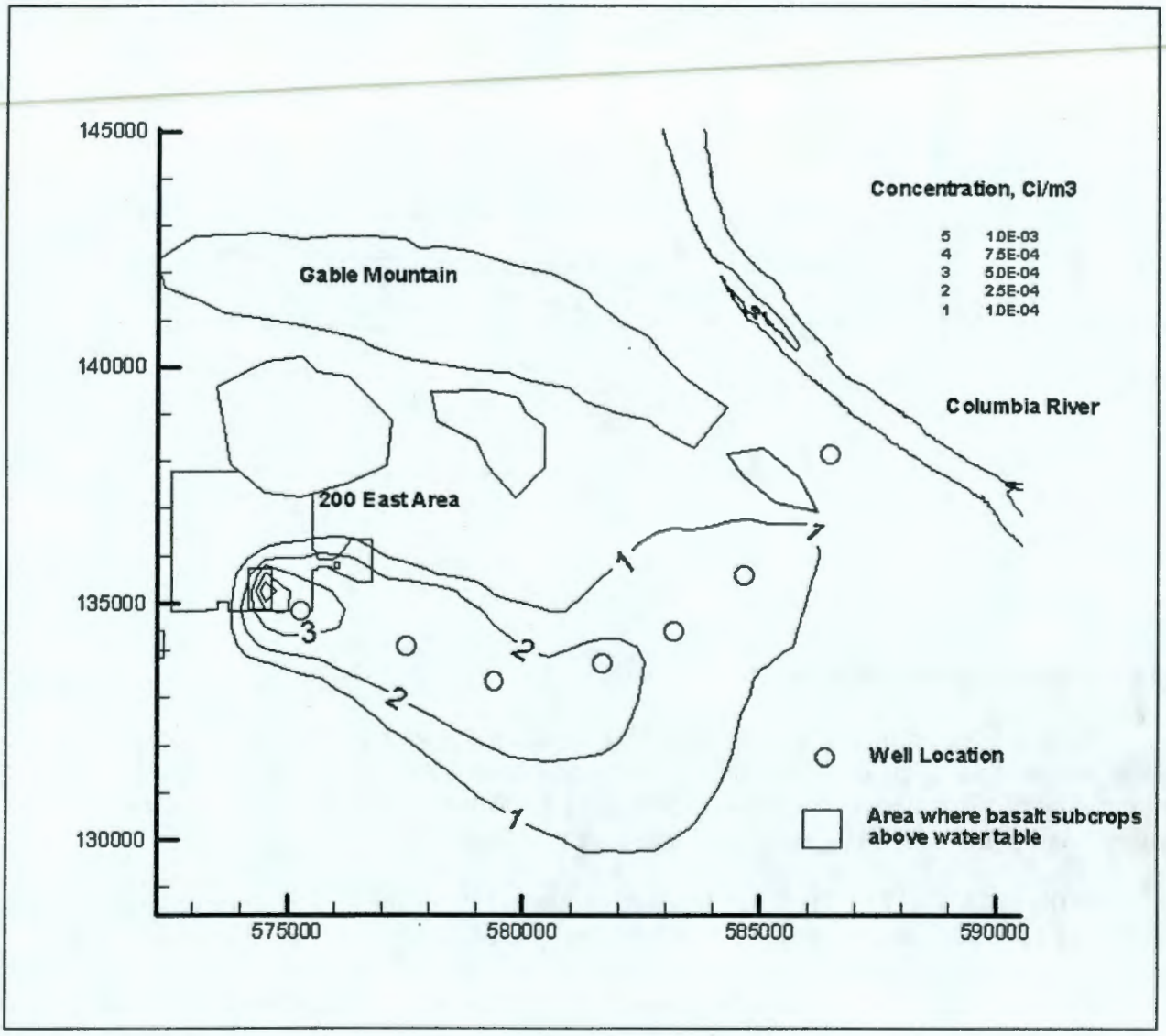
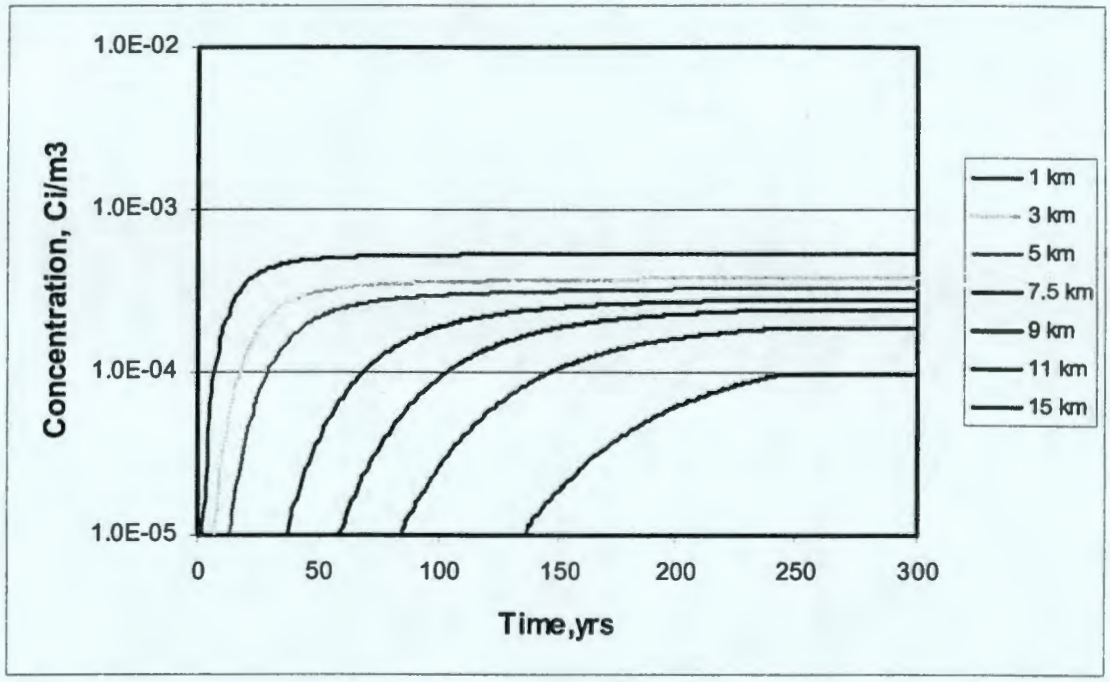


Table 4-2. Well Intercept Factors.

Distance* (km)	Local Model	Hanford Site Model
0.1	1.05E-03	
1.0	7.8E-04	5.4E-04
3.1		3.8E-04
5		3.3E-04
7.6		2.8E-04
9.3		2.4E-04
11.1		1.9E-04
14.8 (river well)		9.8E-05

*Approximate downgradient distances from disposal facilities for well locations shown in Figure 4-15.

Figure 4-16. Time History of Groundwater Concentrations At The Greater Distances.



4.3.6 Base Analysis Case: Conversion into Dose

Once the contaminant concentrations are calculated, it is relatively simple to convert the results into dose using the dosimetry parameters from Section 3.4.7. Table 4-3 displays the estimated impact from the base analysis case for a well 100 m downgradient from the disposal facility. As can be seen, all the calculated levels are quite low.

In the 1998 ILAW performance assessment (Mann 1998a), the most restrictive impact was caused by the drinking water dose from beta- and photon-emitting radionuclides. At 10,000 years, this dose was calculated to be 2.0 mrem in a year resulting mainly from ^{99}Tc (75 percent) and ^{79}Se (20 percent). This performance assessment shows much lower numbers at 10,000 years (0.010 mrem/y). The highest value calculated for the beta/photon drinking water dose is 0.013 mrem/year at about 76,500 years. Table 4-4 shows the major contributions at 1,000 years and 10,000 years to the estimated beta and photon drinking water dose at a well 100 m downgradient from the disposal facility. Figure 4-17 shows the time dependence. In this assessment, ^{99}Tc is still the most important radionuclide, contributing approximately 58 percent of the dose at 1,000 and 10,000 years. However, the next most important radionuclide is ^{129}I, which contributes approximately 42 percent at 1,000 and 10,000 years. The switch of selenium and iodine is a direct result of site-specific data increasing selenium's K_d from 0. to 4.0 mL/g and decreasing iodine's K_d from 3.0 to 0 mL/g (see Section 3.4.3.3).

The all-pathways dose, like the drinking water dose, is small. Table 4-5 shows the major contributions at 1,000 years and 10,000 years to the estimated all-pathways dose at a well 100 m downgradient from the disposal facility. Figure 4-18 shows the time dependence. The major contaminants of concern at short times are ^{99}Tc and ^{129}I, with the uranium isotopes and ^{237}Np

becoming more important at approximately 10,000 years. The maximum all-pathways dose was calculated at about 100,000 years, having a value of 0.6 mrem/year.

As in the 1998 ILAW performance assessment, the concentration of alpha emitters is small because they are not mobile in the Hanford formation soils. Table 4-6 shows the major contributions at 10,000 years to the estimated concentration of alpha emitters at a well 100 m downgradient from the disposal facility. Figure 4-19 shows the time dependence. Again the concentration of alpha emitters is small. No significant concentration of alpha-emitting radionuclides occurs before 3,000 years. The uranium isotopes and ^{237}Np are the major drivers.

As one progresses from the disposal site, the impacts of alpha emitters become less. Table 4-7 displays the impacts at 100 m and 1000 m from the disposal facility, as well as at a well just before the groundwater mixes with the Columbia River. As noted in the groundwater section (Section 4.3.5), the impacts do not drop quickly for distances close to the disposal facility because of the presence of the ancestral Columbia River channel. At the Columbia River, the impacts are reduced by a factor of 9 because of radionuclide dispersion in the groundwater.

The impact from chemicals in the waste form is very small. Table 4-8 displays the estimated impacts at 1,000 and 10,000 years to the groundwater. Because all chemicals are treated as mobile, they have a similar time dependence as the ^{99}Tc curve shown in Figure 4-18. Similarly, Table 4-9 displays the estimated impacts to the surface water calculated at a well just before entering the Columbia River. Again, the estimated impacts to both the groundwater and surface waters are at least a factor of 1,000 lower than the performance goals.

Table 4-3. Estimated Impacts from the Base Analysis Case for a Well 100 m Downgradient from the Disposal Facility.

Type of Impact	Value at		Performance Objective
	1000 years*	10,000 years	
Dose (mrem in a year) from beta- and photon-emitting radionuclides in drinking water	0.000021	0.0102	4.0
All-pathways dose (mrem in a year)	0.000078	0.070	25.0
Concentration (pCi/L) of alpha-emitting radionuclides	1.0×10^{-16}	0.034	15.0
Radium concentration (pCi/L)	0.0	0.0	3.0
Uranium concentration (pCi/L)	0.0	0.025	----

*Time of compliance is 1,000 years.

Table 4-4. Major Radionuclide Contributors at 1,000 Years and 10,000 Years to the Estimated Beta/Photon Drinking Water Dose at a Well 100 m Downgradient from the Disposal Facility.

Radionuclide	1,000 years*		10,000 years	
	Dose (mrem/yr)	Fraction	Dose (mrem/yr)	Fraction
^{99}Tc	0.0000123	0.58	0.00584	0.57
^{129}I	0.0000088	0.41	0.00434	0.43
Others	0.0000002	0.01	0.00002	0
Total	0.0000213	----	0.0102	-----

*Time of compliance is 1,000 years.

Figure 4-17. Time Dependence of the Estimated Beta/Photon Drinking Water Dose at a Well 100 m Downgradient from the Disposal Facility.

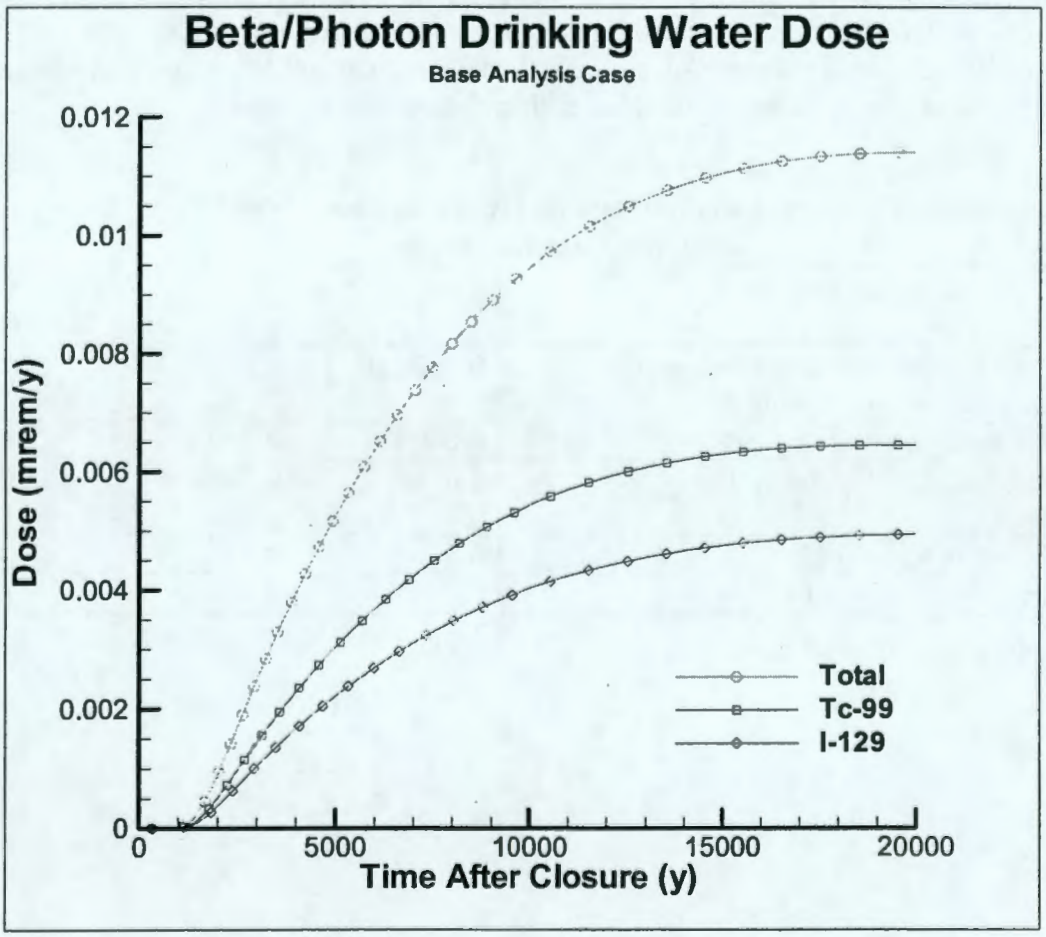


Table 4-5. Major Radionuclide Contributors at 1,000 Years and 10,000 Years to the Estimated All-Pathways Dose at a Well 100 m Downgradient from the Disposal Facility.

Radionuclide	1,000 years*		10,000 years	
	Dose (mrem/yr)	Fraction	Dose (mrem/yr)	Fraction
^{99}Tc	0.0000552	0.71	0.0262	0.37
^{129}I	0.0000224	0.29	0.0110	0.16
^{231}Pa	0.0	0.0	0.00048	0.01
^{233}U	0.0	0.0	0.00310	0.04
^{234}U	0.0	0.0	0.00104	0.01
^{238}U	0.0	0.0	0.00108	0.02
^{237}Np	0.0	0.0	0.0270	0.39
Other	0.0000002	0.0	0.0001	0.0
Total	0.0000778	----	0.0700	-----

*Time of compliance is 1,000 years.

Figure 4-18. Time Dependence of the Estimated All-Pathways Dose at a Well 100 m Downgradient from the Disposal Facility.

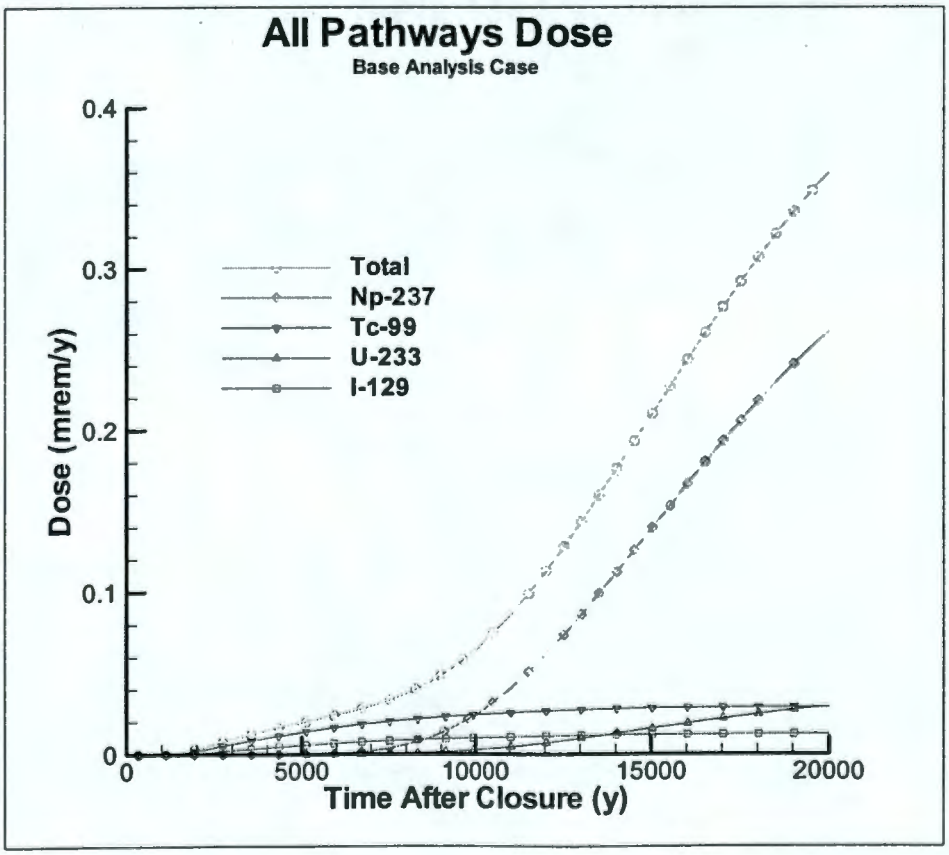


Table 4-6. Major Radionuclide Contributors at 10,000 Years to the Estimated Concentration of Alpha Emitters at a Well 100 m Downgradient from the Disposal Facility. No Significant Concentration Occurs Before 3,000 Years.

Radionuclide	10,000 years	
	Concentration (pCi/L)	Fraction
^{231}Pa	0.0000684	0.00
^{233}U	0.0142	0.42
^{234}U	0.00486	0.14
^{235}U	0.000199	0.01
^{236}U	0.000167	0.0
^{238}U	0.00530	0.16
^{237}Np	0.00910	0.27
Other	0.0000055	0.00
Total	0.0339	-----

Figure 4-19. Time Dependence of the Concentration of Alpha Emitters Dose at a Well 100 m Downgradient from the Disposal Facility.

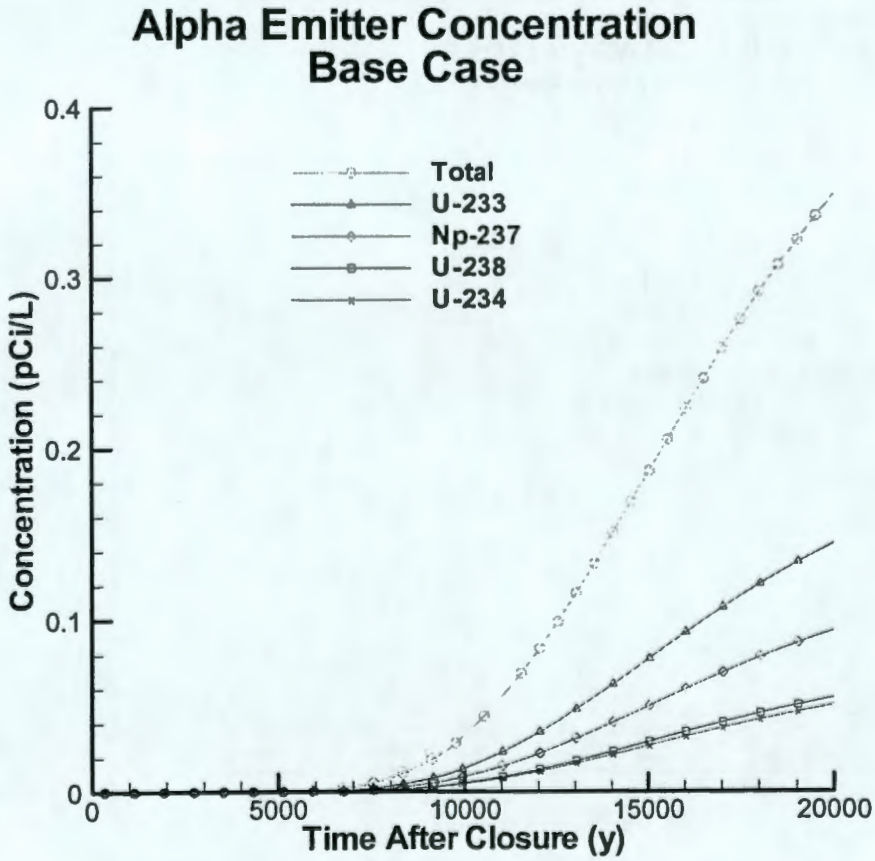


Table 4-7. Estimated Impacts at 1,000 Years from the Base Analysis Case for a Well at Various Locations Downgradient from the Disposal Facility.

Type of Impact	Estimated Impact at 1,000 year* at Well Location		
	100 m	1,000 m	Just Before Entering the Columbia River
Dose (mrem in a year) from beta/photon radionuclides in drinking water	2.13×10^{-5}	1.58×10^{-5}	1.98×10^{-6}
All-pathways dose (mrem in a year)	7.78×10^{-5}	5.78×10^{-5}	7.26×10^{-6}
Concentration (pCi/L) of alpha-emitting radionuclides	1.02×10^{-16}	7.58×10^{-17}	9.99×10^{-18}
Radium concentration (pCi/L)	0.0	0.0	0.0
Uranium concentration (pCi/L)	0.0	0.0	0.0

*Time of compliance is 1,000 years

Table 4.8. Impacts from Chemicals at 1,000 Years and 10,000 Years at a Well 100 m Downgradient from the Disposal Facility.

Chemical	Performance Goal (mg/L) [1]	Estimated Impact (mg/L) @ 1,000 y [2] Nominal Inventory	Estimated Impact (mg/L) @10,000 y	
			Nominal Inventory[4]	Upper Bound Inventory[4]
Ammonia (NH_3)	NA	0.00E+00	0.00E+00	6.63E-04
Antimony (Sb)	0.006	0.00E+00[3]	0.00E+00[3]	0.00E+00[3]
Arsenic (As)	0.00005	4.75E-11	2.33E-08	5.49E-08
Barium (Ba)	1	5.02E-11	2.46E-08	4.48E-06
Beryllium (Be)	0.004	1.66E-12	8.12E-10	2.88E-07
Cadmium (Cd)	0.005	1.70E-10	8.33E-08	1.11E-06
Chlorine (Cl)	250	2.51E-06	1.23E-03	1.24E-03
Chromium (Cr)	0.05	7.40E-07	3.62E-04	8.89E-04
Copper (Cu)	1	1.98E-12	9.70E-10	8.35E-07
Cyanide (CN)	0.2	0.00E+00	0.00E+00	1.44E-04
Fluorine (Fl)	4	2.68E-06	1.31E-03	1.59E-03
Iron (Fe)	0.3	1.21E-07	5.93E-05	1.85E-03
Lead (Pb)	0.05	2.11E-08	1.04E-05	1.11E-04
Manganese (Mn)	0.05	3.73E-08	1.83E-05	2.59E-04
Mercury (Hg)	0.002	5.19E-10	2.54E-07	2.78E-06
Nickel (Ni)	NA	8.24E-08	4.03E-05	2.38E-04
Nitrate (NO_3)	10	0.00E+00	0.00E+00	6.94E-02
Nitrite (NO_2)	1	0.00E+00	0.00E+00	1.67E-02
Nitrite plus Nitrate	10	0.00E+00	0.00E+00	8.61E-02
Selenium (Se)	0.01	1.44E-12	7.05E-10	1.61E-09
Silver (Ag)	0.05	2.92E-10	1.43E-07	4.01E-06

Table 4.8. Impacts from Chemicals at 1,000 Years and 10,000 Years at a Well 100 m Downgradient from the Disposal Facility.

Chemical	Performance Goal (mg/L) [1]	Estimated Impact (mg/L) @ 1,000 y [2] Nominal Inventory	Estimated Impact (mg/L) @10,000 y	
			Nominal Inventory[4]	Upper Bound Inventory[4]
Sulphate (SO_4)	250	9.16E-06	4.48E-03	5.17E-03
Thallium (Tl)	0.002	0.00E+00	0.00E+00	6.72E-05
Uranium (U-total)	NA	4.67E-08	2.29E-05	1.01E-04
Zinc (Zn)	5	5.35E-09	2.62E-06	7.66E-06
1,1,1-Trichlorethane	0.003	0.00E+00	0.00E+00	1.21E-06
1,1,2,2,-Tetrachloroethane	0.005	0.00E+00	0.00E+00	0.00E+00
1,1,2-Trichlorethylene	0.005	0.00E+00	0.00E+00	1.21E-06
1,1,2-Trichloroethane	0.005	0.00E+00	0.00E+00	1.21E-06
1,4-Dichlorobenzene	0.004	0.00E+00	0.00E+00	1.21E-06
Benzene	0.001	0.00E+00	0.00E+00	2.02E-06
Carbon tetrachloride	0.0003	0.00E+00	0.00E+00	1.21E-06
Chloroform	0.007	0.00E+00	0.00E+00	1.21E-06
Dichloromethane (Methylene chloride)	0.005	0.00E+00	0.00E+00	6.07E-06
Ethyl benzene	0.1	0.00E+00	0.00E+00	2.02E-06
N-butyl alcohol	NA	0.00E+00	0.00E+00	5.26E-07
o-Xylene	0.7	0.00E+00	0.00E+00	6.07E-06
Toluene	1	0.00E+00	0.00E+00	2.02E-06

[1]Groundwater performance goal (see Table 1.4).
[2]1,000 years is the time of compliance.
[3]Antimony inventory assumed equal to zero.
[4] Nominal and upper bound inventory are defined in Section 3.2.3.
NA Not Applicable

Table 4-9. Impacts from Chemicals at 1,000 Years and 10,000 Years at a Well Just Before Entering the Columbia River.

Chemical	Performance Goal (mg/L)[1]	Estimated Impact (mg/L) @ 1,000 y [2] Nominal Inventory	Estimated Impact (mg/L) @10,000 y	
			Nominal Inventory[4]	Upper Bound Inventory[4]
Ammonia (NH_3)	4	0.00E+00	0.00E+00	6.94E-05
Antimony (Sb)	0.006	0.00E+00[3]	0.00E+00[3]	0.00E+00[3]
Arsenic (As)	0.05	4.98E-12	2.44E-09	5.75E-09
Barium (Ba)	2	5.26E-12	2.58E-09	4.70E-07
Beryllium (Be)	0.004	1.74E-13	8.51E-11	3.02E-08

Table 4-9. Impacts from Chemicals at 1,000 Years and 10,000 Years at a Well Just Before Entering the Columbia River.

Chemical	Performance Goal (mg/L)[1]	Estimated Impact (mg/L) @ 1,000 y [2] Nominal Inventory	Estimated Impact (mg/L) @10,000 y	
			Nominal Inventory[4]	Upper Bound Inventory[4]
Cadmium (Cd)	0.00077	1.78E-11	8.73E-09	1.16E-07
Chlorine (Cl)	230	2.63E-07	1.29E-04	1.30E-04
Chromium (Cr)	0.011	7.75E-08	3.80E-05	9.31E-05
Copper (Cu)	0.0078	2.07E-13	1.02E-10	8.74E-08
Cyanide (CN)	0.0052	0.00E+00	0.00E+00	1.51E-05
Fluorine (F)	4	2.81E-07	1.38E-04	1.66E-04
Iron (Fe)	NA	1.27E-08	6.21E-06	1.94E-04
Lead (Pb)	0.0015	2.22E-09	1.09E-06	1.16E-05
Manganese (Mn)	NA	3.90E-09	1.91E-06	2.72E-05
Mercury (Hg)	0.000012	5.43E-11	2.66E-08	2.91E-07
Nickel (Ni)	0.115	8.63E-09	4.23E-06	2.49E-05
Nitrate (NO_3)	10	0.00E+00	0.00E+00	7.28E-03
Nitrite (NO_2)	1	0.00E+00	0.00E+00	1.75E-03
Nitrite plus Nitrate	10	0.00E+00	0.00E+00	9.02E-03
Selenium (Se)	0.005	1.51E-13	7.39E-11	1.69E-10
Silver (Ag)	NA	3.06E-11	1.50E-08	4.20E-07
Sulphate (SO_4)	NA	9.59E-07	4.70E-04	5.42E-04
Thallium (Tl)	NA	0.00E+00	0.00E+00	7.04E-06
Uranium (U-total)	NA	4.89E-09	2.40E-06	1.05E-05
Zinc (Zn)	0.072	5.60E-10	2.74E-07	8.02E-07
1,1,1-Trichlorethane	0.2	0.00E+00	0.00E+00	1.27E-07
1,1,2,2,-Tetrachloroethane	0.005	0.00E+00	0.00E+00	0.00E+00
1,1,2-Trichlorethylene	0.005	0.00E+00	0.00E+00	1.27E-07
1,1,2-Trichloroethane	0.005	0.00E+00	0.00E+00	1.27E-07
1,4-Dichlorobenzene	0.075	0.00E+00	0.00E+00	1.27E-07
Benzene	0.005	0.00E+00	0.00E+00	2.12E-07
Carbon tetrachloride	0.005	0.00E+00	0.00E+00	1.27E-07
Chloroform	NA	0.00E+00	0.00E+00	1.27E-07
Dichloromethane (Methylene chloride)	0.005	0.00E+00	0.00E+00	6.36E-07
Ethyl benzene	0.1	0.00E+00	0.00E+00	2.12E-07
N-butyl alcohol	NA	0.00E+00	0.00E+00	5.52E-08
o-Xylene	0.7	0.00E+00	0.00E+00	6.36E-07
Toluene	1	0.00E+00	0.00E+00	2.12E-07

Table 4-9. Impacts from Chemicals at 1,000 Years and 10,000 Years at a Well Just Before Entering the Columbia River.

Chemical	Performance Goal (mg/L)[1]	Estimated Impact (mg/L) @ 1,000 y[2] Nominal Inventory	Estimated Impact (mg/L) @10,000 y	
			Nominal Inventory[4]	Upper Bound Inventory[4]

[1]Surface water performance goal (see Table 1.4)
[2]1,000 years is the time of compliance.
[3]Antimony inventory assumed equal to zero.
[4] Nominal and upper bound inventory are defined in Section 3.2.3.
NA Not applicable

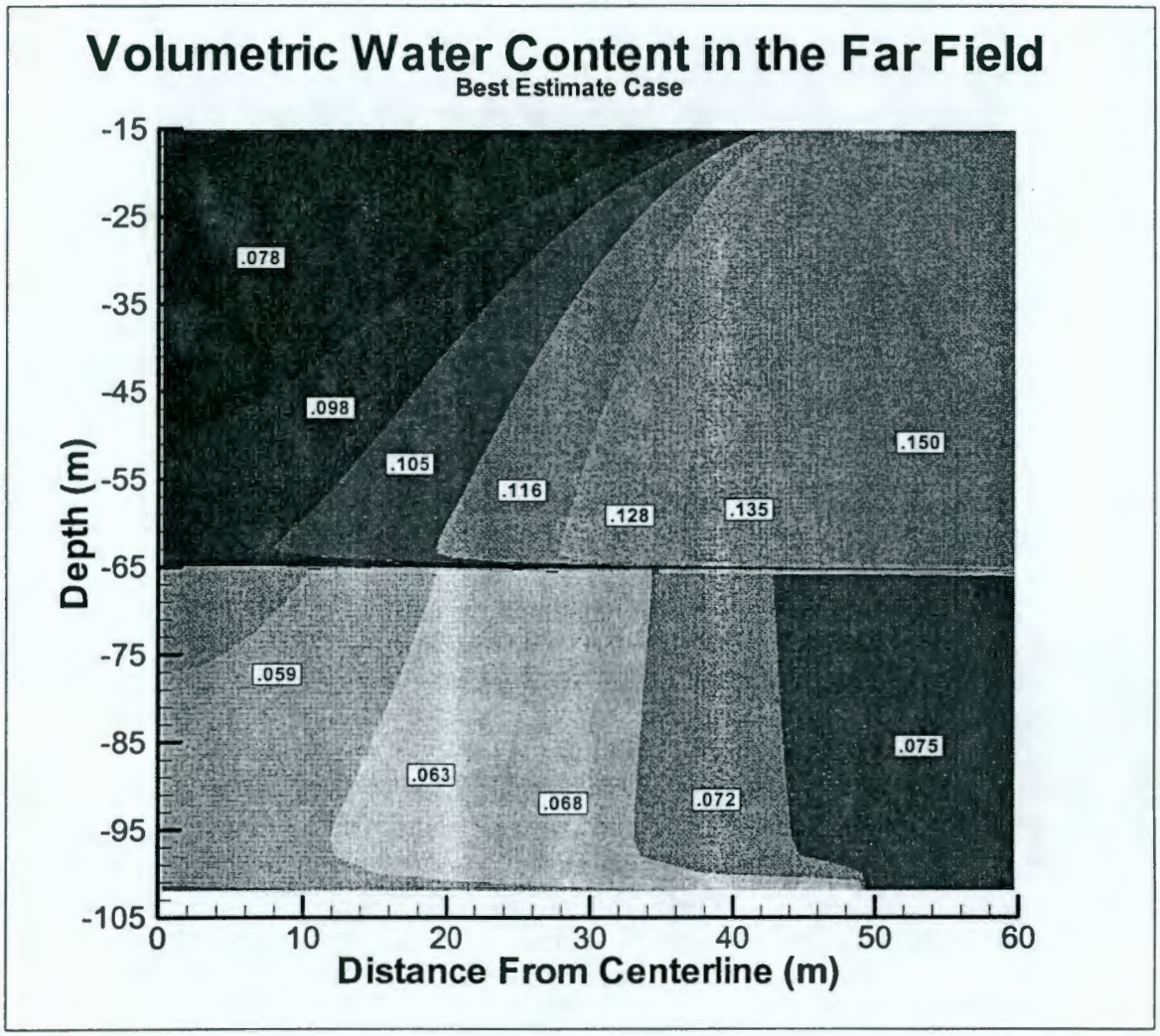
4.3.7 Best Estimate Case: Moisture Flow Into Disposal Facility

The difference between the base analysis case and the best estimate case is the presence of the sand-gravel capillary barrier in the best estimate case. In the base analysis case, this volume was assumed to be sand. This subsurface barrier would divert moisture away from the top of the disposal facility. If the barrier works as well in practice as it does in theory, the results are impressive (see Figure 4-7 where the moisture influx rates for the base analysis case and the best estimate case are shown). Inclusion of the barrier would reduce the influx rate from about 4.2 mm/y to less than 0.01 mm/y.

As seen from Figure 4-6, the moisture flux (or Darcy velocity) greatly decreases with a capillary barrier in place. Directly underneath the barrier, the drop in moisture velocity is over 4 orders of magnitude, resulting in a Darcy velocity of less than 0.003 mm/year. Even at 32 m from the center of the trench (outer edge of waste package region), the Darcy velocity is down by over two orders of magnitude. As expected, at the edge of the cap, the velocity increases to approximately 60 mm/year. This decreased moisture flux within the waste package region results in a much lower contaminant concentrations as seen in Figure 4-7, where the best estimate case is labeled "Cap. Break" for capillary break. For the best estimate case, the moisture velocities inside the disposal are quire small (being less than 0.1 mm/y). Therefore, to reduce computational time the contaminant release rate associated with a recharge rate of 0.1 mm/year was used in these calculations. This assumption leads to conservative estimates for the contaminant flux leaving the disposal facility. It should be noted that, because these calculations do not include water consumption by the glass dissolution reaction, the release rates shown are even lower than presented here.

Figure 4-20 shows the volumetric moisture content in the far field when the capillary barrier is in place. The moistest area (15 percent) remains unchanged as the 50 mm/year recharge at the edge of the facility remains unchanged. However, directly beneath the disposal facility the volumetric moisture drops from 11 percent (base analysis case) to 6 percent.

Figure 4-20. Volumetric Moisture Content With The Sand-Gravel Capillary Barrier Atop The Disposal Facility.



4.3.8 Best Estimate Case: Waste Form Release

Waste form release is calculated for a series of infiltration rates, so that a space-dependent release rate from the disposal facility can be estimated.

As discussed in Section 4.3.3, calculations of the waste form release rate and of the contaminant fluxes out of the disposal facility were performed using a one-dimensional model for various recharge rates. Based on the space-dependent moisture flows calculated in Section 4.3.7, these contaminant fluxes then were translated into a space-dependent contaminant flux out of the disposal facility.

Figure 4-7 (curve labeled "Cap. Break") displays the resulting calculated contaminant concentration out of the disposal facility at 1,000 years for the best estimate case. As can be seen, the stair-step effect caused by the layers of waste packages remains, but the contaminant flux leaving the disposal facility is lower by two orders of magnitude when compared to the base analysis case.

4.3.9 Best Estimate Case: Moisture Flow And Contaminant Transport in the Vadose Zone

This large decrease in contaminant flux continues as the material moves through the vadose zone. As seen from Figure 4-8 (curve labeled "Cap. Break"), the contaminant flux at 10,000 years at its spatial peak is over four orders of magnitude less with a capillary barrier than without one.

As would be expected, the total inventory-normalized contaminant flux entering the groundwater for the best estimate case (Figure 4-21) is much lower than the base analysis case. At 10,000 years, the best estimate case has values over 4 orders of magnitude less. This large difference is because of the lower contaminant flux leaving the disposal facility, as well as the longer travel time in the vadose zone caused by the lower moisture flow. Figure 4-22 compares the contaminant flux leaving the disposal facility (entering the vadose zone) with the contaminant flux entering the groundwater (leaving the vadose zone). Unlike the base analysis case where the vadose travel time is about a thousand years, for the best estimate case, the vadose zone travel time is on the order of a few tens of thousands of years. Also, the contribution of the less mobile contaminants is less when compared to the base analysis case because the contaminant flux for these radionuclides (K_d = 0.6 mL/g) is more than two orders of magnitude less than the mobile contaminants (K_d = 0 mL/g) at 10,000 years (see Figure 4-22).

Figure 4-21. Total Inventory-Normalized, Mobile (K_d = 0 mL/g), Contaminant Flux to the Groundwater for the Base Analysis and Best Estimate Cases as a Function of Time.

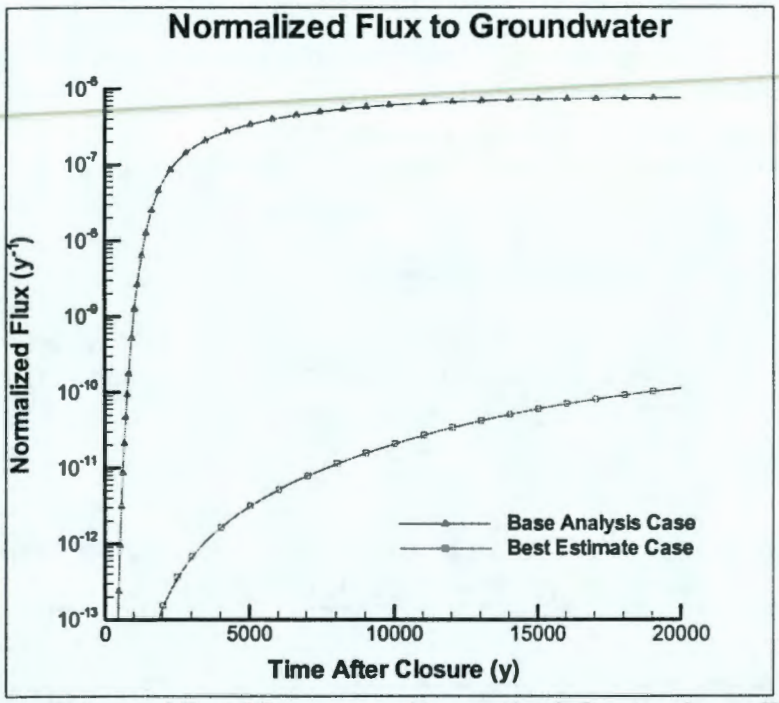
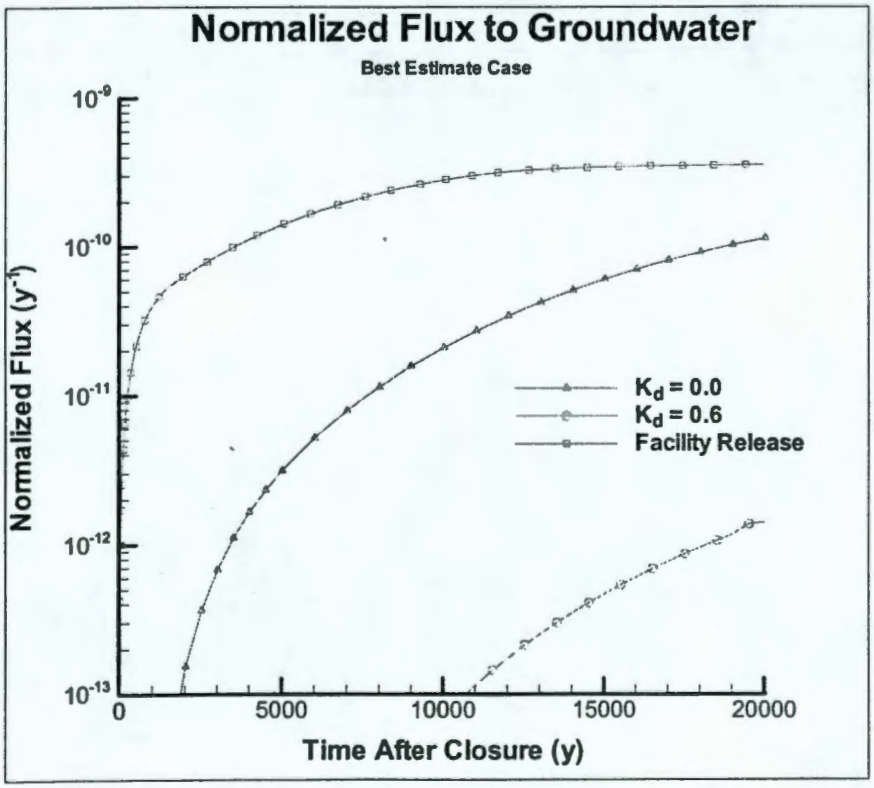


Figure 4-22. Comparison of Total Inventory-Normalized Contaminant Flux Leaving the Disposal Facility and Entering into the Groundwater for the Best Estimate Case for K_d = 0 and 0.6 mL/g as a Function of Time.



4.3.10 Best Estimate Case: Moisture Flow And Contaminant Transport in Groundwater

The presence of a sand-gravel capillary barrier does not affect groundwater flow as the amount of recharge with or without the capillary barrier is insignificant at the disposal location compared to the regional moisture influx that actually controls the magnitude of groundwater flow. Therefore the results from Section 4.3.5 are used without any changes.

4.3.11 Best Estimate Case: Conversion into Dose

The presence of the subsurface sand-gravel capillary barrier greatly reduces moisture flow into the disposal facility, which greatly reduces contaminant release from the waste form and greatly increases travel time to the groundwater. Table 4-10 presents the impacts from this best-estimate case.

Table 4-10. Estimated Impacts From The Best Estimate Case For A Well 100 Meters Downgradient From The Disposal Facility.

Type of Impact	Value at		Performance Objective
	1000 years*	10,000 years	
Dose (mrem in a year) from beta/photon radionuclides in drinking water	4.7×10^{-11}	3.5×10^{-7}	4.0
All-pathways dose (mrem in a year)	1.7×10^{-10}	1.3×10^{-6}	25.0
Concentration (pCi/L) of alpha-emitting radionuclides	2.2×10^{-22}	5.0×10^{-8}	15.0
Radium concentration (pCi/L)	0.0	0.0	3.0
Uranium concentration (pCi/L)	0.0	3.7×10^{-8}	----

*Time of Compliance is 1,000 years.

4.4 SENSITIVITY CASES: MOISTURE FLOW INTO DISPOSAL FACILITY

Table 4-11 shows the large number of two-dimensional simulations that were run. These simulations varied recharge rates, barrier design, and the type of disposal facility (trench versus vault). The resulting Darcy rates for key cases are shown in Figure 4-6.

The effect of recharge rates is as expected. The greater the rate, the more moist the soil and the greater the moisture flux. Without a subsurface capillary barrier, there is little difference in the moisture velocity fluxes. The large differences in all-pathways dose result from the strong dependence of contaminant release from the waste form on moisture content.

As shown in Section 4.3.7, the presence of the sand-gravel capillary barrier is significant. It reduces the amount of moisture entering the disposal facility by about four orders of magnitude in the driest areas. Even at the edge of the disposal facility, the moisture flux is still reduced by two orders of magnitude. The moisture flux does increase outside the facility, rising to approximately 60 mm/year outside the barrier. However, this high velocity region is away from the waste and has minimal effect on the waste form release.

If the surface barrier and the capillary barrier are shortened by 5 m so it extends only 12 m past the last waste package, the velocity past that package increases by about a factor of 5 compared to the full barrier case. The estimated impacts for both the beta-photon drinking water and all-pathways doses are approximately a factor of 20 larger for the shortened capillary break analysis when compared to the best estimate case. At 10,000 years, the estimated beta-photon drinking water and all-pathways doses for the shortened capillary break are approximately a factor of three greater than the doses estimated for the best estimate case (see Table 4-10).

To investigate the importance of subsidence, a local failure of the capillary barrier is postulated. A 1-meter subsidence in the capillary barrier refers to a trough created in the capillary break 10 meters from the trench centerline. The trough is created by displacing the surface barrier downward 0.3 m over a 1-meter distance (the third dimension being ignored in this two-dimensional calculation). The selection of a 0.3 m downward displacement corresponds to a complete collapse of the free volume (5%) in the stack height of four waste packages. In this subsidence case, the local recharge increases to 11.2 mm/year over a relatively small region. Moisture is diverted into the region just underneath the failure. The higher moisture increases the contaminant release from the waste form, leading to larger estimated impacts. The total effect on the all-pathways dose of having 1 m of the subsurface sand-capillary barrier fail is to increase the estimated impact from 1.7×10^{-10} mrem/year to 2.0×10^{-8} mrem/year at 1,000 years and from 1.3×10^{-6} mrem/year to 1.5×10^{-2} mrem/year at 10,000 years when compared to the best estimate case.

Table 4-11. Summary of Near-Field Hydraulic Sensitivity Calculations.

Description	Basic Model	Velocity Range [a] (mm/y)	All-Pathways Dose (mrem/y)[b]	
			1,000 years [c]	10,000 years
Base analysis case	Trench	4.1 - 4.3	7.8×10^{-5}	7.0×10^{-2}
Rupert sand (0.9 mm/year)	Trench	0.87 - 0.95	1.38×10^{-8}	5.22×10^{-4}
Low recharge rate (0.1 mm/year)	Trench	0.098 - 0.11	2.54×10^{-11}	2.64×10^{-6}
High recharge (50 mm/year)	Trench	49. - 51.	3.99×10^{-1}	1.61
Full barrier (best estimate case)[d]	Trench	1.2×10^{-4} - 8.5×10^{-3}	1.71×10^{-10}	1.31×10^{-6}
Full barrier –High recharge rate (50 mm/year)	Trench	1.0×10^{-3} - 1.0×10^{-1}	7.3×10^{-9}	3.76×10^{-6}
Base case – sand backfill	Trench	4.1 - 4.2	1.64×10^{-3}	2.35×10^{-1}
No side slope effect	Trench	4.2 - 4.2	4.01×10^{-6}	4.48×10^{-2}
Vertical hydraulic barrier	Trench	1.2×10^{-4} - 8.3×10^{-3}	0.00	7.51×10^{-7}
Capillary barrier failure[e]	Trench	0.0094 - 11.1	1.95×10^{-8}	1.45×10^{-2}
Shorter capillary barrier[f]	Trench	1.2×10^{-4} - 1.5×10^{-2}	3.12×10^{-9}	4.41×10^{-6}
Burbank sand (4.2 mm/year)	Vault	4.1 - 4.8	2.21×10^{-2}	7.38×10^{-1}

[a] Over region of waste packages
[b] Performance objective is 25 mrem/year
[c] Time of compliance is 1,000 years
[d] Labeled as "Cap. Break" in Figure 4-6
[e] Labeled as "Subsidence" in Figure 4-6
[f] Labeled as "Short. Cap. Break" in Figure 4-6

4.5 SENSITIVITY CASES: WASTE FORM RELEASE

4.5.1 Overview

The 1998 ILAW performance assessment (Mann 1998a) showed that the estimated impacts strongly depended on the rate of waste form release. Therefore, a large number of sensitivity runs were made to determine the sensitivity of the results to various inputs and assumptions. Table 4-12 provides a summary of the results. It should be noted that relative changes in the waste form release rate translate directly into changes in impacts (with a time delay corresponding to the vadose zone travel time), because the waste form release rates dominate the time structure of the problem. All of these sensitivity cases, but one, were simulated in one-dimension.

4.5.2 Forward Rate of Reaction

Forcing glass dissolution to be at the maximum rate increases the estimated release rate from the disposal facility by a factor of 9.

As glass dissolves, silicon is released into the surrounding fluid. As the $SiO_2(aq)$ concentration in the fluid increases, the rate of glass dissolution decreases. However, the exact description of how this rate slows is still a matter of research. To obtain a bounding case, the glass dissolution model was run assuming that the glass dissolution is always at the maximum rate consistent with temperature (which is unvarying) and pH, the so-called forward rate of reaction. Figure 4-23 compares the inventory-normalized contaminant flux to the vadose zone when the forward rate is used.

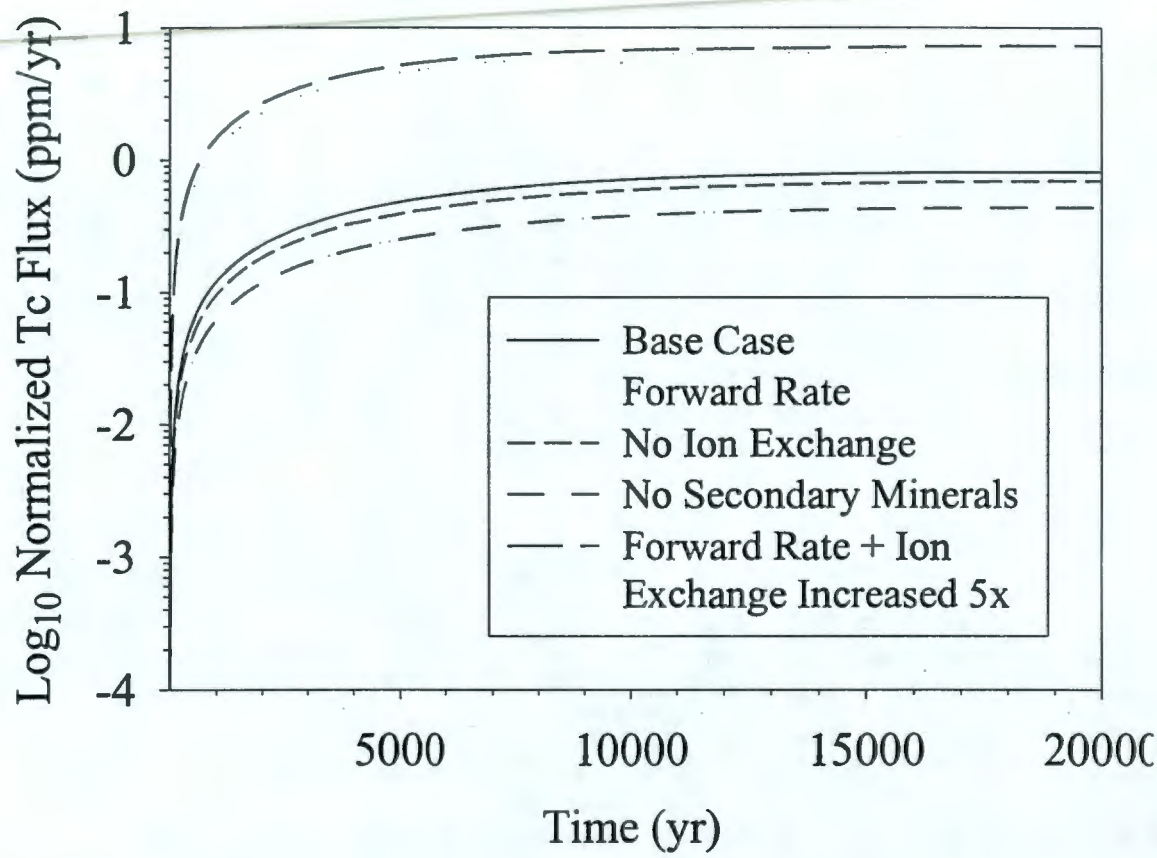
As expected, the release rate to the vadose zone greatly increases (by a factor of ~9) over the base case if the forward rate is used. This is because of a combined higher intrinsic rate of glass dissolution and a higher pH in the disposal facility.

It should be noted that the *White Paper Updating the Conclusions of the 1998 ILAW Performance Assessment* (Mann/Puigh 2000b) only used forward rate calculations, which explains why the estimated impacts in that document are higher than those in this PA.

Table 4-12. Summary of Waste Form Sensitivity Calculations. The flux ratio translates into the impact ratio (except for a time delay corresponding to the vadose zone travel time).

Section	Description	Basic Model	Flux Ratio @ 1,000 years	Flux Ratio @ 10,000 years	Reason Higher or Lower than Base Case
4.3.3	Base case - 4.2 mm/year recharge	Trench	Base case	Base case	
4.5.2	Forward rate, 4.2 mm/year recharge	Trench	9.93	8.71	Forward rate
4.5.3	Assume no ion exchange	Trench	0.79	0.83	Lower pH
4.5.3	Assume no secondary phase formation	Trench	0.52	0.53	Higher Si concentration
4.5.3	Increase ion exchange rate by 5 times while also using forward rate	Trench	12.17	9.51	Forward rate, higher pH
4.5.4	0.1 mm/year recharge	Trench	0.0004	0.0004	Lower recharge
4.5.4	0.5 mm/year recharge rate	Trench	0.0052	0.0041	Lower recharge
4.5.4	0.9 mm/year recharge rate	Trench	0.016	0.016	Lower recharge
4.5.4	10 mm/year recharge rate	Trench	8.44	2.64	Higher recharge
4.5.4	50 mm/year recharge rate	Trench	20.9	3.40	Higher recharge
4.5.5.1	Change filler material in trench to sand	Trench	9.13	1.62	Water content/diffusion higher in sand
4.5.5.2	Include steel in waste packages	Trench	1.03	1.02	Steel corrosion increases pH
4.5.5.3	Add conditioning layer at top	Trench	0.99	0.92	Higher Si concentration
4.5.6	4.2 mm/year recharge	Vault	8.55	2.67	Vault, glass packed closer, higher pH
4.5.6	Replace concrete everywhere with backfill material	Vault	9.11	2.48	Vault, glass packed closer, higher pH
4.5.6	0.9 mm/year recharge rate	Vault	0.75	0.20	Vault, lower recharge
4.5.6	0.1 mm/year recharge	Vault	0.055	ran to 1800 years	Vault, lower recharge
4.5.7	Increase Waste Loading / Alternate Glass Formulation	Trench	164	76	Higher release rate, no dependence on Si
4.5.8	Increase diffusion for all aqueous species by a factor of 10	Trench	0.06	0.011	Lower pH
4.5.8	Replace Tc with U	Trench	1.03	1.00	Slight soddyite precipitation
4.5.8	Full two-dimensional simulation	Trench	2.65	ran to 2,000 years	Lower water flux through glass
4.5.8	Extend contaminant release model to groundwater	Trench	0.93	1.09	Position of lower boundary has small affect on release rate

Figure 4-23. Inventory-Normalized Contaminant Flux to the Vadose Zone for Different Glass Dissolution Models.



4.5.3 Glass Dissolution Rate

Neglecting the sodium ion exchange or the creation of secondary minerals slightly reduces the estimated release rate.

The main reaction that causes glass to dissolve in water is breaking the bridging oxygen bonds that hold the glass matrix together. However, other reactions also go into the waste form release calculations. The two most important are the sodium ion exchange reaction and the formation of secondary minerals that affect the silicon concentration.

In the sodium ion exchange reaction, hydrogen ions in the water replace the sodium in the glass. This reaction thus increases the amount of sodium in the water and the pH. Because of the high sodium content of the glass and because this reaction has a lower activation energy than the matrix destruction reaction, it can be significant under ILAW disposal conditions. The effect of neglecting the sodium ion exchange reaction is shown in Figure 4-23. If no ion exchange reaction is present, the glass dissolution is reduced, but not significantly. If the ion exchange rate

for the forward rate simulation is increased five times, the result is a slight increase in the overall glass dissolution rate.

As the amount of silicon builds up in the fluid, it can precipitate out as a secondary mineral, thus reducing the concentration. This causes the glass to dissolve more quickly. As seen in Figure 4-23, the effect is real and if the precipitation rate is rapid enough to reduce the silica in solution to very low values, glass dissolution rates can accelerate back up to the forward rate of reaction.

Secondary minerals also can trap important contaminants. Research for the ILAW PA program has shown that 80 percent of the technetium can be trapped in such materials. However, although the experiments are continuing (see section 7.8.2 for a more complete discussion of future work on waste forms), we do not know yet whether such trapping is a short-term effect or will last the hundreds and thousands of years necessary to affect these calculations. Therefore, this performance assessment assumes that none of the important contaminants are trapped in secondary phases over the long term. Once better data are obtained for the secondary minerals, the effects will be incorporated in future performance assessments.

4.5.4 Effect of Infiltration Rate

> The waste release rate from the disposal facility increases as the infiltration rate increases. However, the release rate saturates for infiltration rates of about 50 mm/year.

Moisture plays an important role in glass dissolution because it carries the reaction products away from the dissolving glass and into the vadose zone. Two effects are important: one is the transport of the contaminants once they are released and the other is the effect on the local chemistry.

Figure 4-4 shows the estimated contaminant flux from the disposal facility into the vadose zone. Higher recharge rates flushes dilute water through the system, lowering the silicon concentration in the fluid surrounding the glass and increasing the glass dissolution rate. However, the higher recharge rates have a point of diminishing returns. The diluting effect of higher recharge also restricts increases in pH. Such increases in pH would increase the glass dissolution rate. As seen from Figure 4-4, above 10 mm/y, the release rates from the disposal facility increase little with increased recharge rate.

These calculations neglect the fact that water is needed to dissolve the glass (see equation 3.8 in Section 3.4.4.2). However, at the 0.1 mm/year recharge rate (the lowest rate calculated), the glass dissolution rate would be limited to 0.015g/(m^2 day), which still is 7 times the forward rate at 15°C and pH=9 for LAWABP1 glass. Thus ignoring water consumption is a valid strategy.

4.5.5 Effect of Materials Surrounding ILAW Packages

> The material surrounding the waste form can have a significant effect for times less than 5,000 years. However, neither the waste container nor the preconditioning layer has much affect.

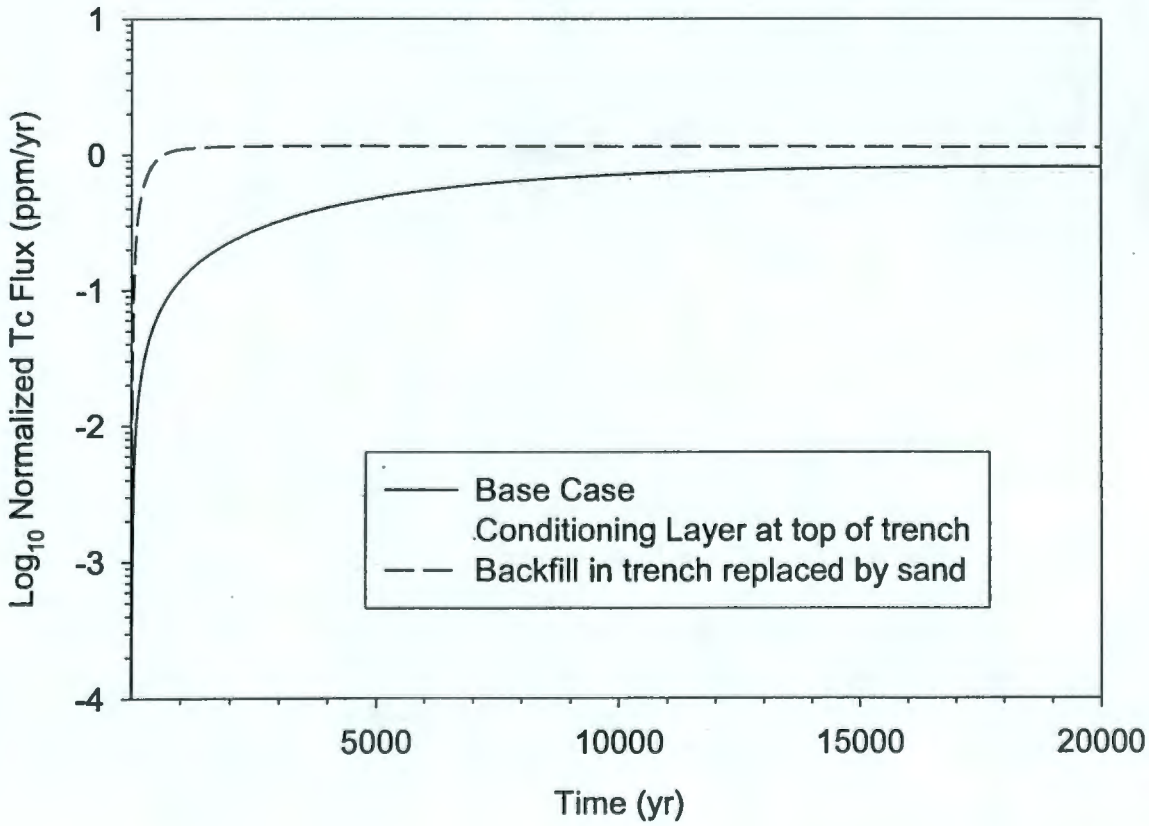
Many design decisions still need to be made concerning the disposal facility. Some of

these will affect the glass dissolution rate. This section investigates the effect of the following:

- Different filler material between the ILAW packages
- Steel containers
- Conditioner layering above the disposal facility.

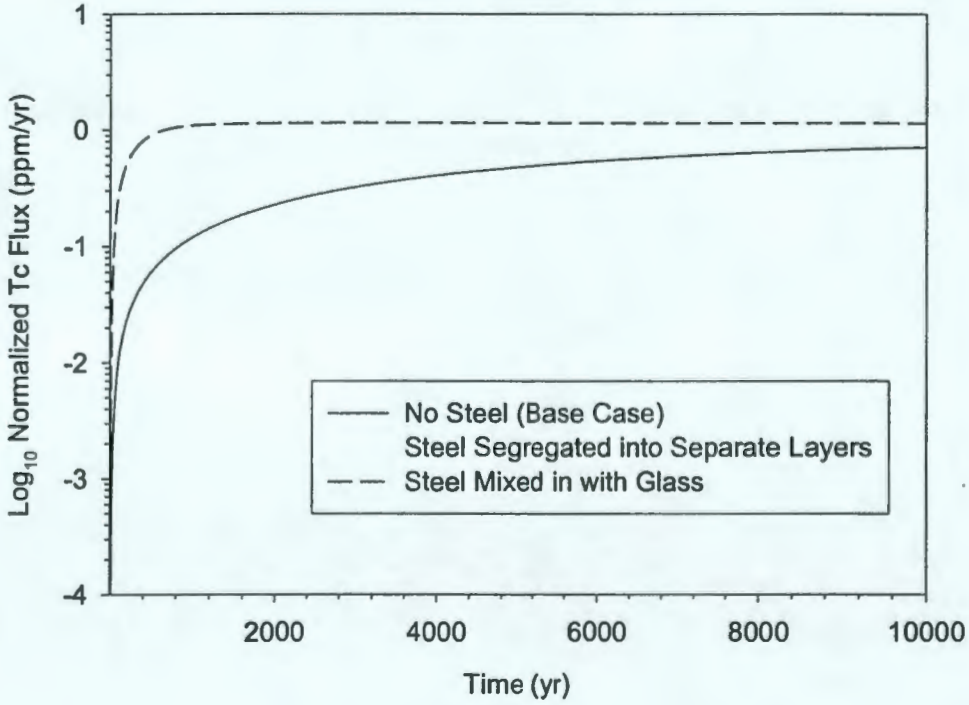
4.5.5.1 Effect of Filler Material on Glass Dissolution. The ILAW packages will be separated by filler material. For the base analysis case, the hydraulic properties of backfill material (i.e., excavated material) were used. Figure 4-24 shows the effect of replacing such material with sand. Although over long periods the dissolution rates approach each other, at short times (less than 5,000 years), they are significantly different. The difference is caused by the sands having a lower moisture content, and thus a higher linear pore velocity at a recharge rate of 4.2 mm/y. The higher pore velocity in the intervening sand layers allows Tc to be transported downward to the bottom of the facility more rapidly. The moisture content within the glass layers is very similar whether the filler material is backfill or sand, and thus the long-term dissolution rates are very similar for these two cases.

Figure 4-24. Contaminant Flux to the Vadose Zone Assuming Different Surrounding Materials.



4.5.5.2 Steel Containers. The base analysis case and subsequent calculations have ignored that the ILAW waste package is actually glass surrounded by a stainless steel container. Including the stainless steel in the reaction network increases the pH slightly, because the corrosion reaction consumes H^+. However, as shown in Figure 4-25, the effect is small when the steel containers are modeled correctly as layers separate from the glass waste. If the steel is completely mixed with the glass in one material zone, it raises the local pH and enhances the precipitation of the iron-bearing clay nontronite, which lowers the silicon concentration and significantly increases the glass dissolution rate. This underscores the importance of using at least a one-dimensional model to simulate the layered waste form, as opposed to a "zero-dimensional" reaction kinetics code, such as EQ3/6, where all materials in the waste form would be mixed together.

Figure 4-25. Contaminant Flux to the Vadose Zone Assuming Different Surrounding Materials.



Presently, the plan is to have the ILAW in 304L stainless steel packages. Because this large amount of steel could add potential contaminants to the groundwater, a separate simulation was run to model the release of chromium and nickel to the vadose zone. Based on limited knowledge of steel corrosion rates gained in the Yucca Mountain Project (Cloke et al. 1997), a constant rate of steel corrosion of 6.9×10^{-14} mol cm^{-2} s^{-1} was assumed. By assuming this rate, the stainless steel corrodes away (i.e. is oxidized) entirely within about 1,100 years. Corrosion and oxidation of the 304L is also assumed to release Cr directly to the aqueous phase as Cr(VI), i.e. the important intermediate step of oxidizing Cr(III) to Cr(VI) was assumed to occur faster than the rate of Cr(III) release from steel corrosion. Because Cr(VI) is present almost entirely as the

soluble and mobile CrO_4^{2-} ion, the approximately 10^7 kg of chromium in the stainless steel would also be released in about 1,100 years. In contrast, the solution concentrations of Ni are constrained by precipitation of theophrastite [$Ni(OH)_2$], resulting in negligibly small Ni release to the vadose zone.

4.5.5.3 Conditioning Layer. The amount of silicon in the fluid surrounding the glass is important. Some preconceptual designs for the closure cover had a layer of silica-bearing material just above the disposal facility to precondition the water. Using crushed soda bottles was considered, but because of cost and because its higher surface area than the waste glass would make the crushed glass dissolve too quickly, their use was not considered further. However, these calculations considered the use of 1 m- thick silica sand just above the glass region. However, as seen from Figure 4-24, the effect is quite small.

4.5.6 Effect of Type of Disposal Facility

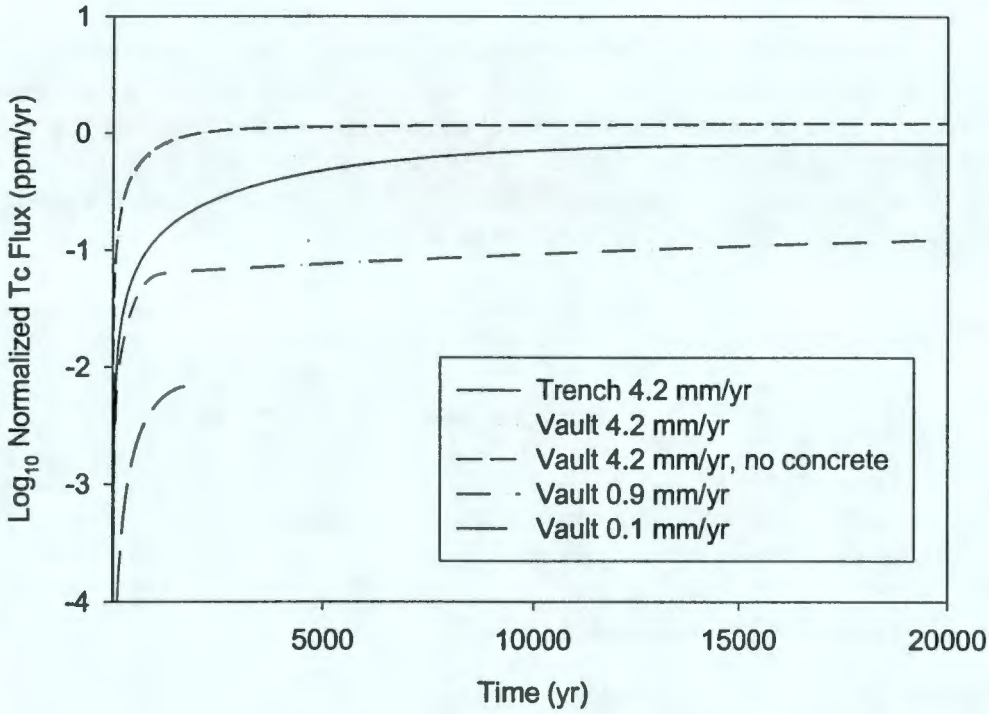
The release rate from the concrete vault is higher (about 3 or more for times less than 5,000 years) than the release rate from the trench.

The 1998 ILAW performance assessment (Mann 1998a) assumed that the ILAW packages would be disposed in underground concrete vaults. Although such concrete vaults are not now part of the baseline, the project is responsible for four unused concrete vaults (the unused grout vaults). Therefore, calculations were performed using these vaults not only to provide a bridge from the 1998 ILAW PA results, but to allow their future use if needed.

The results of the concrete vault simulations are shown in Figure 4-26. For the 4.2 mm/year recharge rate case, the vault has a higher concentration flux to the vadose zone than the trench, particularly at early times. Note that if the concrete is replaced with backfill, the vault values are unchanged. This suggests that the vault performs more poorly than the trench because of the dense waste configuration in the vault rather than because of chemical effects as suggested in Mann/Puigh (2000b).

As in the trench simulations, assuming lower recharge rates results in significantly lower contaminant fluxes to the vadose zone. The 0.1-mm/year recharge rate simulation was unable to progress past 1,800 years because of high concentrations in the vault, which were close to the maximum allowable concentrations of 2.1 mol/kg. For these lower recharge rate simulations, the trench design again performs better, being over an order of magnitude better at 1,000 years.

Figure 4-26. Contaminant Flux to the Vadose Zone for Vault Simulations.



4.5.7 Effect of Waste Loading and Other Glass Compositions

The base analysis case uses a sodium waste loading of 20 weight percent. Increasing the Na_2O loading in LAW glasses has several impacts that can affect the long-term dissolution rate of the product. First, sodium is a glass "network breaker." Adding sodium to silicate glasses depolymerizes the glass, making its structure less interconnected and, so, generally less durable when contacted by water. Second, increased sodium content may make the glass more susceptible to alkali ion exchange reactions. The net affect of ion exchange is to raise the pH of water percolating through the disposal system, thereby increasing the glass dissolution rate. Lastly, increasing the Na_2O content tends to expand the stability field and rate of zeolitic alteration phase formation as the glass reacts with water. The formation of zeolite (a class of hydrous aluminum silicate minerals) can decrease the amount of dissolved silica, thus causing dissolution rate excursions, sometimes all the way back to the forward rate of reaction. Thus, higher sodium loading will likely shrink the composition region from which acceptable LAW glasses can be formulated.

There is only a limited experimental base for LAW glasses at other than 20 weight percent Na_2O loading. From the limited data in Figure 4-27, a simple linear regression gives a slope of $10^{0.2x}$, where x is the mass percent Na_2O. Consequently, a 5 percent increase in Na_2O loading would increase the glass corrosion rate by approximately ten times. However, to conduct a more detailed analysis, laboratory experiments with HLP-31 glass (see Vienna 2000),

which has a Na_2O loading of 23 weight percent, were performed to establish the necessary input parameters for STORM simulations. The results (see Figure 4-28) show a very unusual glass corrosion behavior in that the dissolution rate is apparently unaffected by an increasing concentration of silicon in the aqueous phase, up to near saturation with respect to amorphous silica. Also, the congruent release of Na and B indicates that little or no Na ion exchange is occurring, despite the higher waste loading. Finally, the forward rate of reaction for this glass is about ten times larger than for LAWABP1 glass at the same temperature and pH. The reasons for this unusual behavior may be related to microscale phase separation as a result of the heat treatment used to simulate canister cooling. In any event, to model the performance of this glass in the disposal system, the glass dissolution rate was not allowed to decrease as the Si concentration in the disposal system pore water increased. No experimental data were available on the pH-dependence of the dissolution rate so it was assumed that the reaction rate increased according to $10^{0.4*pH}$, identical to the power law determined for LAWABP1 glass.

Figure 4-27. VHT Corrosion Rate as a Function of Waste Loading (from Vienna [2000]).

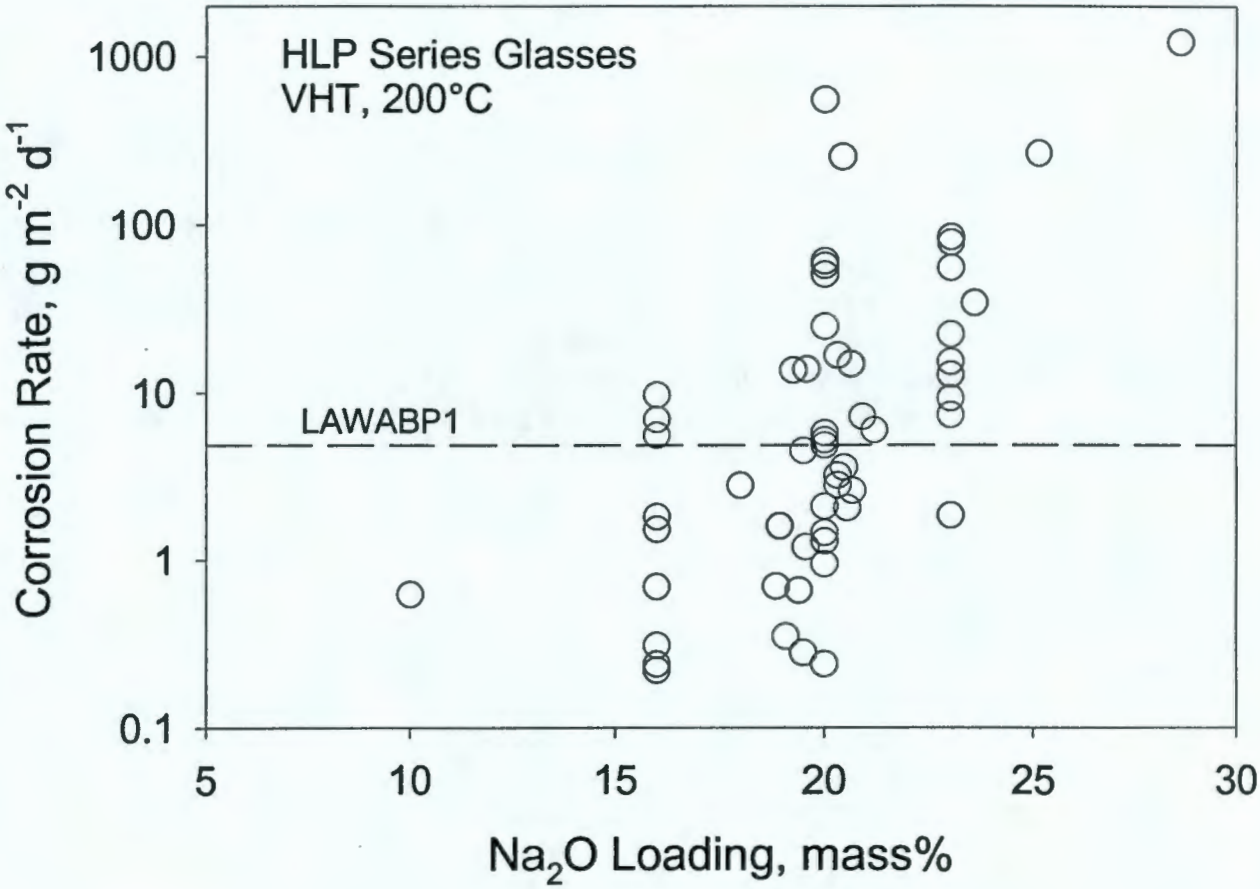
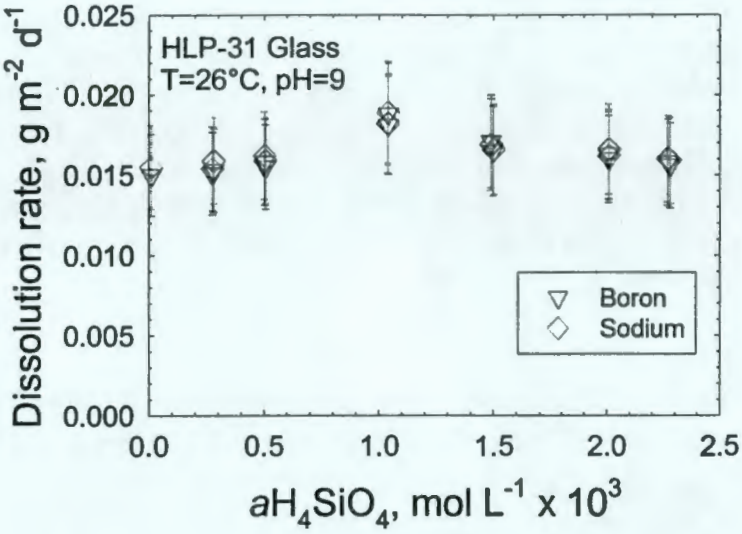


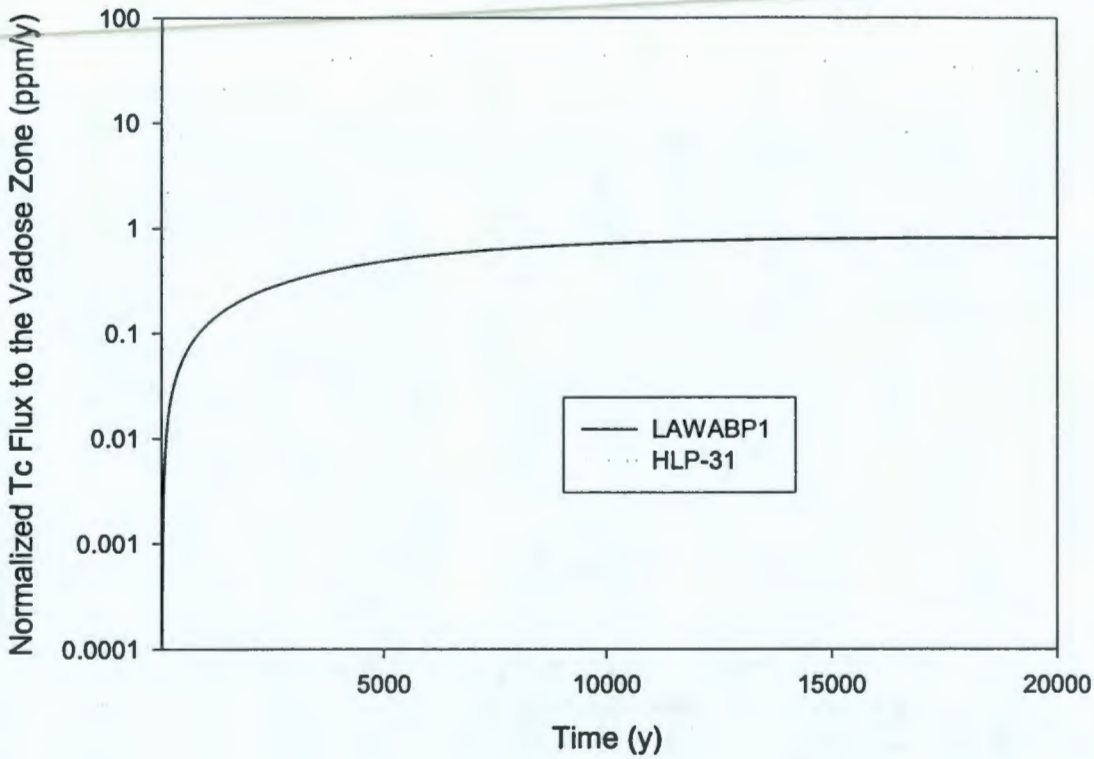
Figure 4-28. Dependence of HLP-31 Glass Dissolution Rate on Concentration of Silicic Acid.



The calculated inventory-normalized contaminant flux to the vadose zone for HLP-31 glass is 75 to 164 times larger (Table 4.12) than that for the base case simulation with LAWABP1 glass, as shown in Figure 4-29. The higher forward reaction rate of HLP-31 glass generates a higher calculated pH in the glass layers as compared with LAWABP1 glass. The combined effect of these factors, and the fact that the rate does not diminish with increasing Si concentration, increases the total release rate much more than the forward rate difference between the glasses alone would indicate. The decline in release rate after 5,000 years occurs because of the decreasing surface area of HLP-31 glass, which changed very little for the slower dissolving LAWABP1 glass over the same time period.

The STORM simulations show the strong sensitivity of release rates to the durability of the glass. Available testing data indicate that Na_2O loadings of 20 to 25 weight percent might be achieved and still produce glasses that will have acceptable long-term performance. VHT testing shows several glasses at 23 weight percent Na_2O loading with a corrosion rate that is nearly as good as LAWABP1 glass (Vienna 2000). Only very limited data at waste loadings above 25 weight percent Na_2O are available. Although it may be possible to formulate acceptable glasses at this waste loading, the acceptable glass composition region will be much smaller than is observed at waste loadings of 20 weight percent Na_2O and lower. Additional studies are needed if waste loadings approaching 25 weight percent Na_2O are desired. Major increases in waste loading on the order of 50 to 100 percent (30 to 40 weight percent Na_2O) are probably not possible with silicate-based glasses. A different glass-forming system, such as the phosphate system, would need to be considered. However, changing to a different glass forming system would also require a different melter design, flow sheet, etc. Also, non-silicate waste glasses have received almost no attention since the early 1980's. Consequently, a significant research and development effort would be required to evaluate long-term performance issues with these glasses.

Figure 4-29. Inventory-Normalized Contaminant Flux to the Vadose Zone for Different Glass Formulations.



4.5.8 Effect of Waste Package Design

The waste form release sensitivity cases thus far have been based upon a cubic waste package with edge dimension of 1.4 m (Puigh 1999). Recent planning for ILAW disposal (DOE/ORP 2000c) has identified a right-circular cylindrical geometry with 1.2 m outside diameter and 2.3 m height as the preferred container configuration. Assuming 85% of the container's available volume is filled with the glass waste form, the height of glass within one container would be approximately 2 m. Current planning (DOE/ORP 2000a) would have three layers of waste packages stacked within an RH trench with dimensions similar to those discussed in Section 2.4. This would result in a maximum glass stack height of 5.9 m.

The vault sensitivity case used a waste stacking height of 7.2 m. Because the principal effect of waste stacking height is to increase the pH of the fluid exiting the bottom of stack (from reaction with the glass), the 7.2 m height of the vault sensitivity case should provide an upper bound of ^{99}Tc release for the 5.9 m stacking height associated with the new container design. The base analysis case was calculated for a waste stacking height of 4.8 m of glass. Therefore, the release from the RH trench using the new canister design is anticipated to lie between the release rates for the base case and the vault case for a given infiltration rate. From Tables 4-1 and 4-12, the anticipated release rate from the RH trench using the new canister design for an infiltration rate of 4.2 mm/y is assumed to be between 0.012 and 1.0 ppm/y at 1,000 years after

facility closure and between 0.71 and 1.9 ppm/y at 10,000 years after facility closure. These estimates neglect the additional dilution that will occur because of the slightly lower volume of glass per package (2.2 m^3 versus 2.3 m^3 for the cubic container) and the larger volume of backfill per cubic meter of glass, which results from less efficient container packing.

4.5.9 Other Calculations

Some other calculations were performed to determine the sensitivity to parameter and model choices. These included the following:

- Increasing the value of the diffusion coefficient
- Modeling the contaminant as uranium
- Extending the reactive transport calculations through the vadose zone to groundwater
- Performing a two-dimensional calculation inside the disposal facility
- Performing a simulation of the "bathtub effect," where the lower boundary of the trench becomes impermeable to water flow, causing the trench to become saturated with water.

Because the moisture content within the disposal facility is so low, advective transport also is low. Under these conditions, diffusion of the contaminants is important. Increasing the aqueous diffusion coefficient for all species from 5×10^{-9} m^2/second (base analysis case) to 5×10^{-8} m^2/second significantly increases the transport of species, increasing the mixing of the aqueous products of glass dissolution. This results in a significantly lower pH in the glass layers and hence a dissolution between 10 and 100 times lower than the base case, depending on the time.

The base analysis case modeled technetium as the main contaminant and did not consider secondary phases. A sensitivity case was run treating uranium as the main contaminant and explicitly treating the secondary phases known to trap uranium. Three uranium aqueous species were considered: $UO_2(CO_3)_2^{2-}$, $UO_2(CO_3)_3^{4-}$, and $UO_2(OH)_2(aq)$. Two secondary minerals were considered: weeksite, $K_2(UO_2)_2Si_6O_{15}{:}4H_2O$, and soddyite, $(UO_2)_2(SiO_4){:}2HO_2$. Only soddyite precipitated, and it did so in miniscule amounts for a very short period. Because of the slow release rate from the glass, steady rate of mass transport through the system, and strong carbonate complexes associated with U(vi), the pore fluid remained undersaturated with respect to weeksite and soddyite and the uranium remained dissolved in the aqueous phase. Hence, the inventory-normalized fluxes for total uranium are identical to those predicted for technetium.

A sensitivity case was run that is identical to the base case, except that the Hanford sand layer was assumed to extend 88 m to the water table. Thus, the total depth of this extended source-term model was 103 m for the extended grid, as opposed to 15 m for the base case simulation. This simulation covered 5,000 years. Simulated pH (Figure 4-30) and TcO_4^- (Figure 4-31) concentrations for the upper 15 m of the extended simulation are very similar to those of

the base case until after 10,000 years, when the effect of the lower water table boundary begins to propagate up the profile. Even so, the technetium release rate is very similar for the base case and the extended simulation, the flux from the extended simulation being 7 percent lower at 1,000 years and 9 percent higher at 10,000 years than the base case. Longitudinal diffusion causes a decrease in the concentrations of aqueous species across the depth of the Hanford sand. pH decreases slightly from the top of the Hanford sand to the water table (Figure 4-30). TcO_4^- decreases from the top of the Hanford Sand to the water table by a factor of 3 at 2,000 years. By 20,000 years, the profile is closer to steady state, and the decrease in technetium is only 9 percent from the top of the Hanford sand to the water table.

A sensitivity case was run that considers the consequences of the trench liner being impermeable to flow, thus causing the trench to become saturated with water. The trench was initially considered to be saturated with water, with no flow across the bottom boundary. Simulated TcO_4^- (Figure 4-32) and total water mass (Figure 4-33) concentrations are similar to those predicted by the base case. However, because the amount of water in the profile is so much higher than in the base case, the amount of technetium released is much higher.

Figure 4-30. pH for RH Trench Simulation Extended to Water Table. (See Figure 3-13 for Location of Boundaries Between Material Zones and Material Names.)

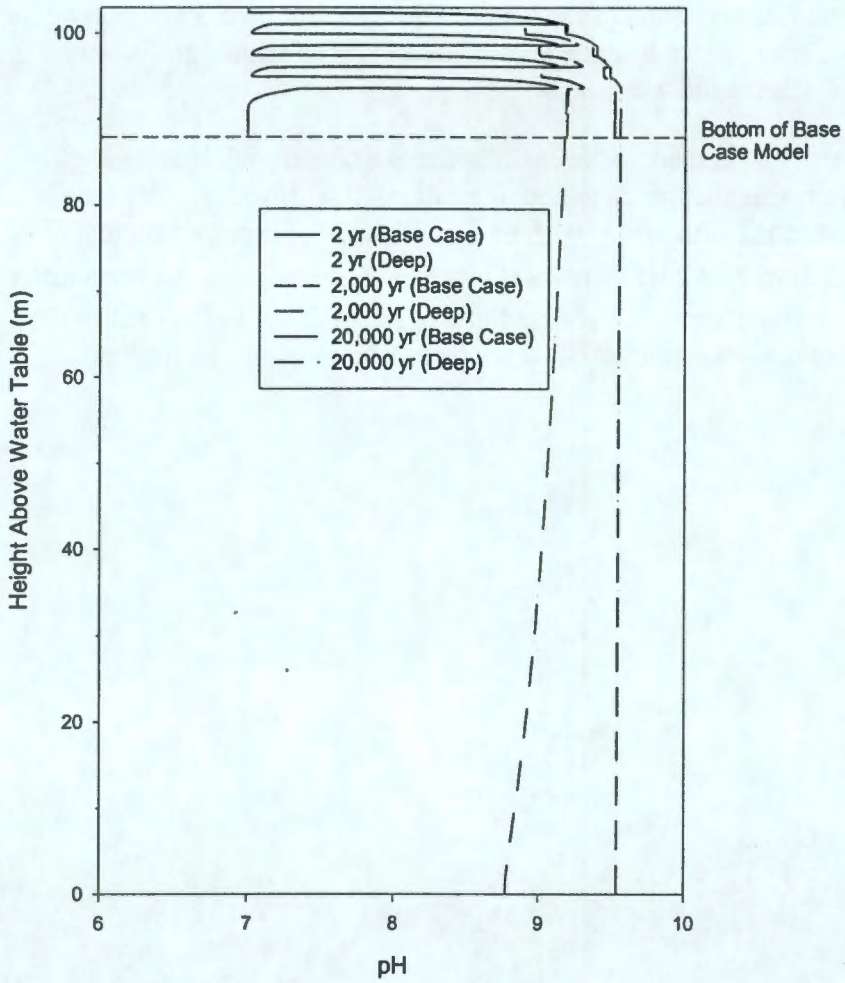


Figure 4-31. TcO_4^- for RH Trench Simulation Extended to Water Table. (See Figure 3-13 for Location of Boundaries Between Material Zones and Material Names.)

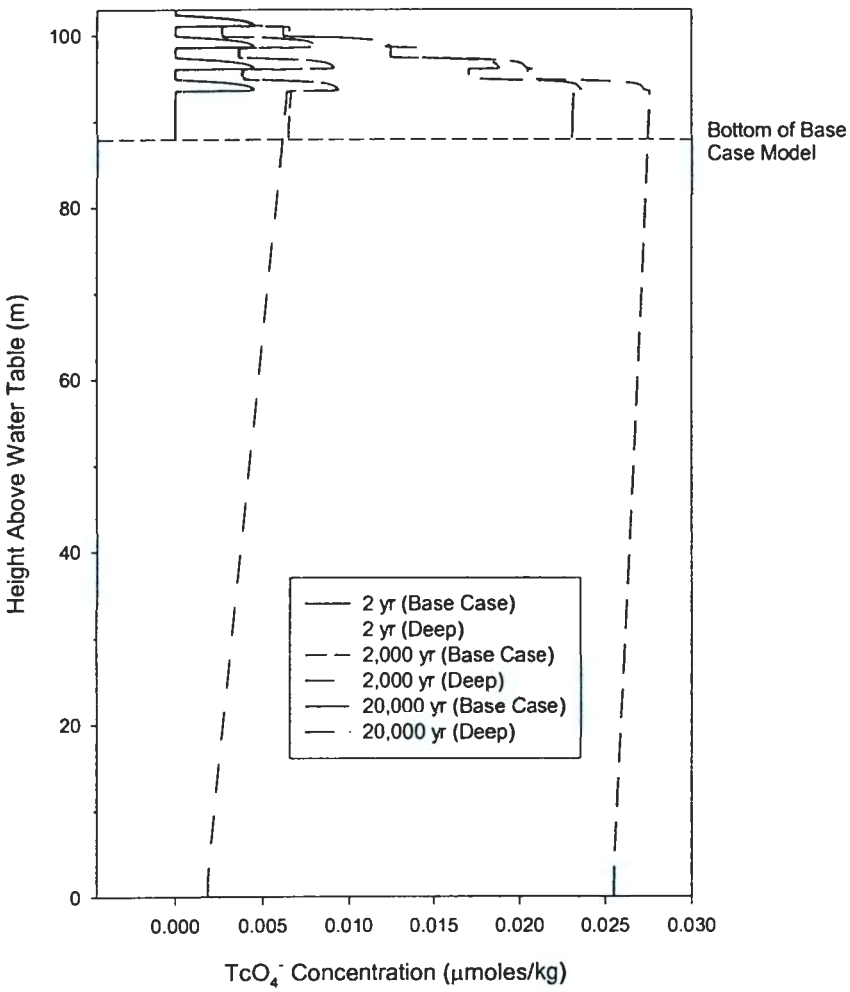


Figure 4-32. TcO_4^- For Fully-Saturated RH Trench Simulation With No Flow ("Bathtub Effect").

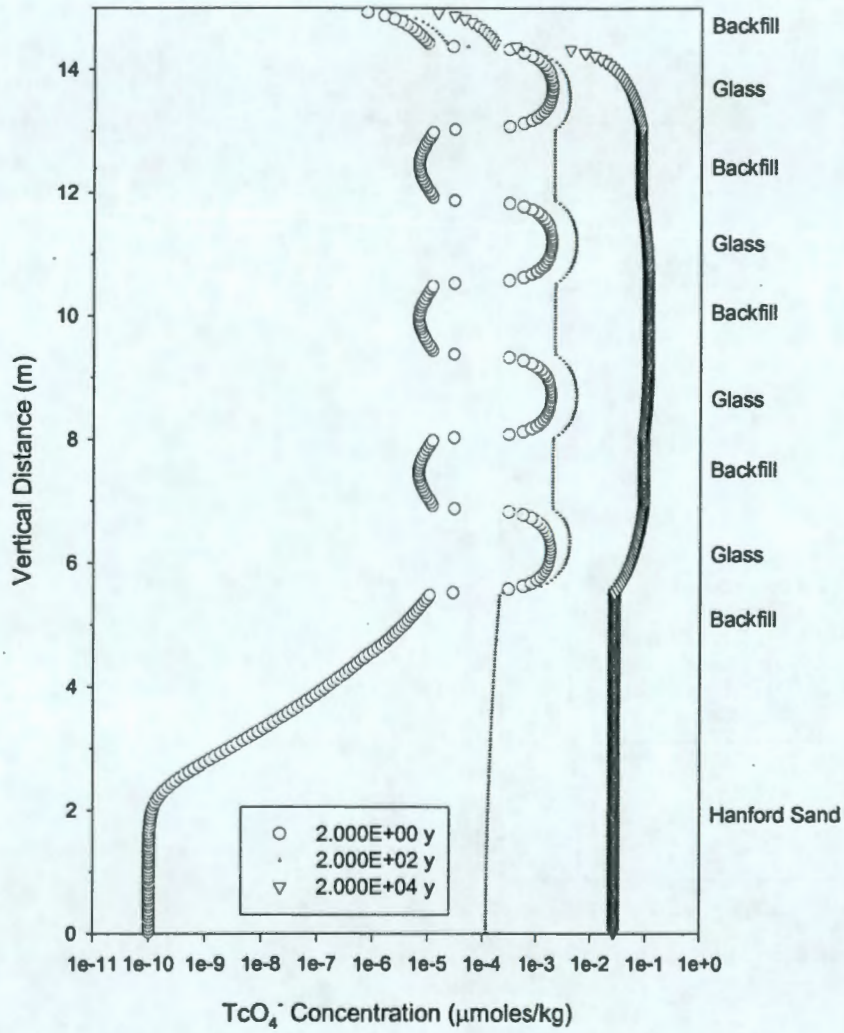
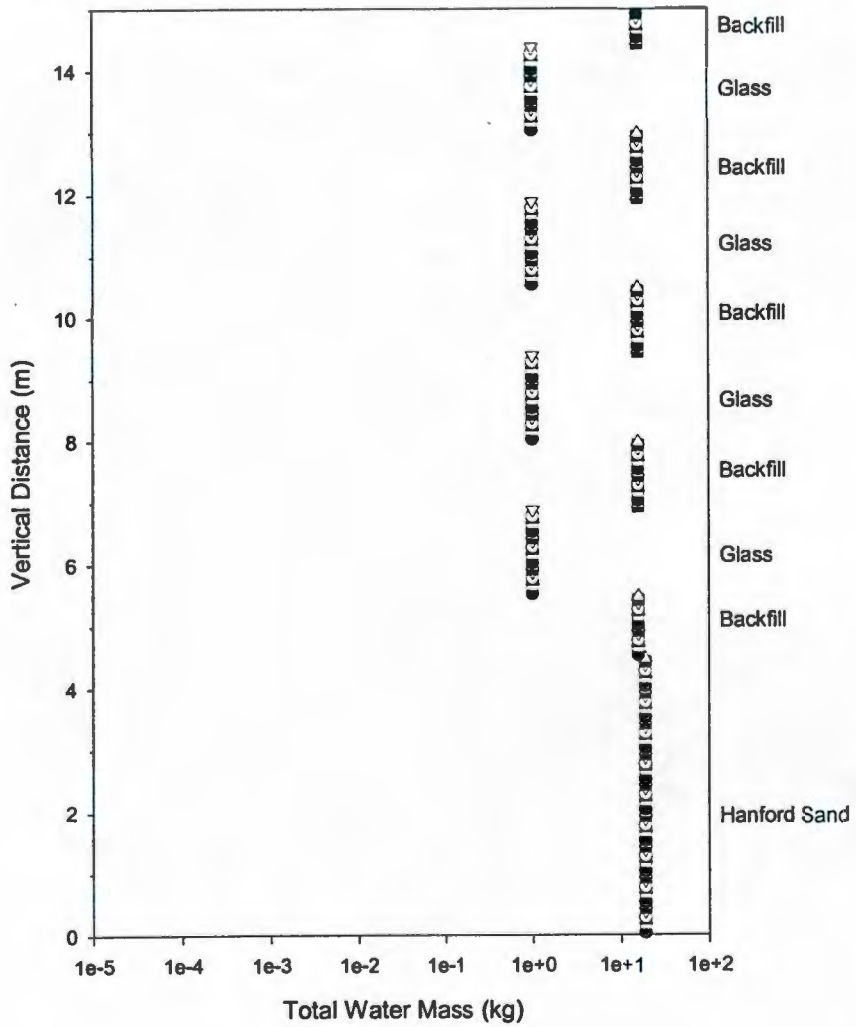


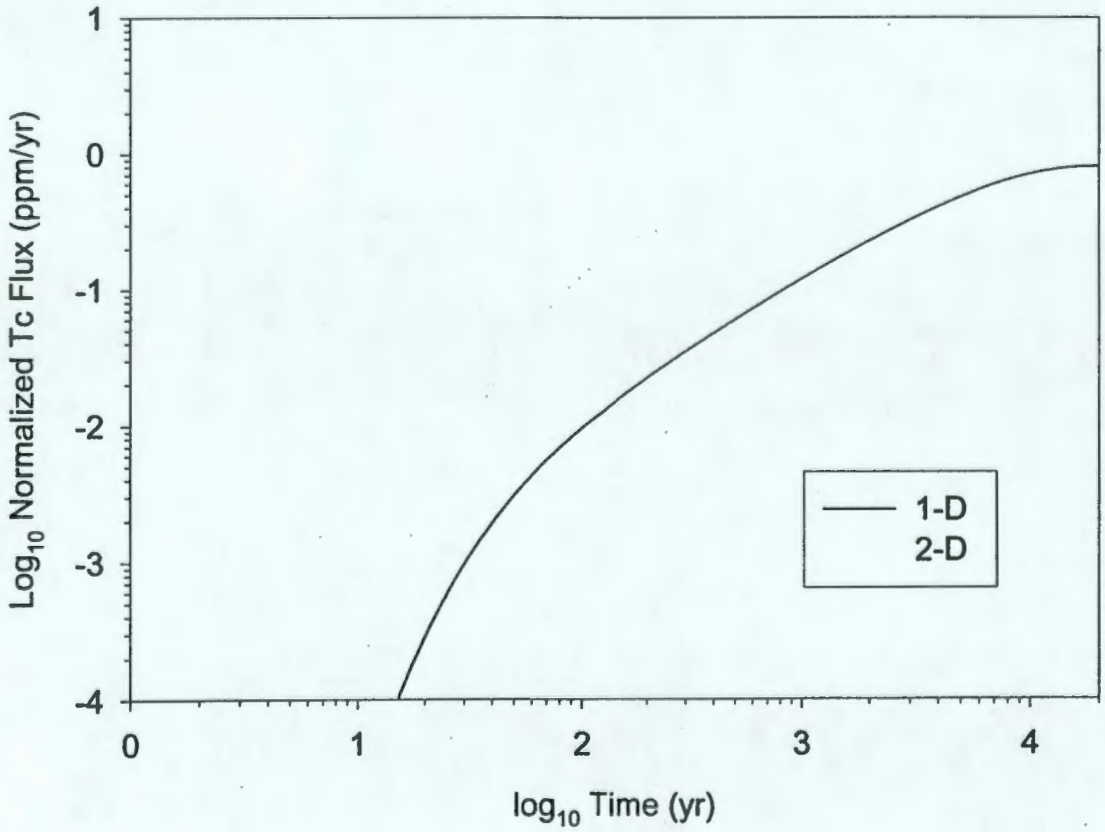
Figure 4-33. Total Water Mass Per Node. (Each Node Has A Volume Of 0.05 m^3.) For Fully-Saturated RH Trench Simulation With No Flow ("Bathtub Effect").



A full two-dimensional simulation of the trench scenario was developed for comparison to the one-dimensional simulation used as the base case. The two-dimensional simulation reaches steady state earlier and shows a lower maximum (or steady state) technetium release rate than the one-dimensional simulation (Figure 4-34). In the two-dimensional simulations, water flows around the glass waste packages. This lowers the water content in, and water flux through the waste packages, and increases the technetium concentrations in the waste packages. The glass dissolution rate for these simulations is highest on the edges of the glass layers. This is where the pH of the pore water is highest and the SiO_2(aq) concentrations are lowest.

This two-dimensional simulation required over a month of computation to reach a simulation time of 2,000 years, while the one-dimensional simulation required only 5.3 days to reach a simulation time of 100,000 years. The results of this comparison support the contention that the predictions of one-dimensional simulations are conservative with respect to, and not dramatically different from, those of the two-dimensional simulations.

Figure 4-34. Inventory-Normalized Contaminant Flux to the Vadose Zone for One-Dimensional (1D) and Two-Dimensional (2D) Simulations. Note that the 2D simulations only go to 2,000 years.



4.6 SENSITIVITY CASES: VADOSE ZONE MOISTURE FLOW AND CONTAMINANT TRANSPORT

4.6.1 Overview

This section discusses the calculations of the moisture flow and contaminant transport in the far-field vadose zone. The major items that will be covered are as follows:

- Recharge
- Vadose zone properties
- Waste form release time dependence.

Table 4-13 summaries the results of the simulations by providing the travel times through the vadose zone. Travel time is defined as the time at which the contaminant flux to the aquifer for the mobile contaminants (K_d = 0 mL/g) is 50 percent of its maximum value, which may be at the maximum time for which calculations were performed (i.e. 20,000 years).

4.6.2 Recharge

As can be seen from the results of the base analysis (Section 4. 3.4) and the best estimate (Section 4.3.9), the recharge rate is important in determining when the contaminant flux enters the groundwater. Figure 4-35 shows the total contaminant flux entering the groundwater normalized to a unit inventory in the waste form for four different recharge rates, assuming no capillary barrier is placed over the disposal facility. Because these calculations used the corresponding waste form release rates, the results reach different asymptotic values. Table 4-13 shows the estimated impacts to the all-pathways doses for different recharge rates. The estimated impacts of recharge rates on the other performance objectives are provided in Appendix D, Table D-1.

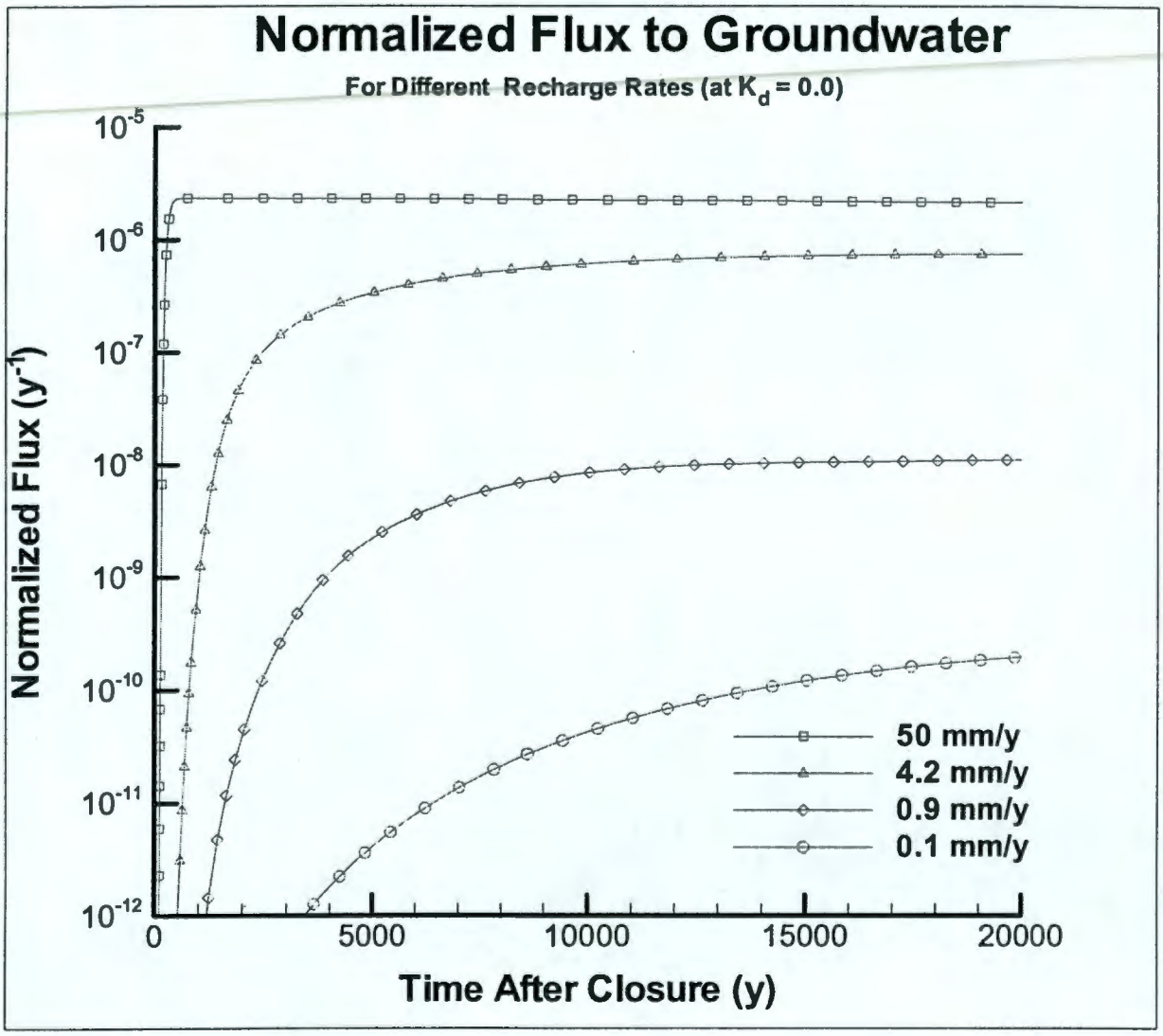
The recharge rate of 50 mm/year is consistent with an estimated recharge rate for irrigation farming above the disposal site. The corresponding beta / gamma drinking water dose and alpha concentrations associated with a well 100 meters downgradient from the disposal site would increase to 0.037 mrem/year and 1.52 pCi/L at 10,000 years after facility closure. The alpha concentration and the all-pathway dose are more impacted than the beta/photon drinking water dose because the contaminants with K_d= 0.6 mL/g are transported more rapidly through the vadose zone when compared to the base analysis case. These estimated impacts for irrigation directly above the disposal site are far below the performance objectives at 1,000 and 10,000 years after facility closure. Moreover, if irrigation were to occur in the region surrounding the ILAW disposal site, then the directional flow of the groundwater would change and the resulting impacts would be lower than the case described above due to additional dilution in the groundwater (see section 4.7.5).

Table 4-13. Travel Times and All-Pathways Doses for the Various Vadose Zone Sensitivity Cases.

Description	Travel Time (y)	All-Pathways Dose (mrem/y)[a]	
		1,000 years[b]	10,000 years
Base case - 4.2 mm/year	5,400	7.78×10^{-5}	7.00×10^{-2}
0.1 mm/year	13,500	2.54×10^{-11}	2.64×10^{-6}
0.9 mm/year	7,300	1.38×10^{-8}	5.22×10^{-4}
50. mm/year	<200	3.99×10^{-1}	1.61
0.1 mm/year for 500 years, then 4.2 mm/year[c]	5,900	6.65×10^{-8}	6.06×10^{-2}
Isotropic flow field	5,300	4.83×10^{-5}	4.96×10^{-2}
10x increase in diffusion	5,300	1.16×10^{-4}	8.05×10^{-2}
2 times increase in dispersion	5,300	1.37×10^{-4}	8.73×10^{-2}
Vadose zone is all sand	5,700	1.96×10^{-5}	3.91×10^{-2}
Vadose zone in all gravel	5,000	3.80×10^{-4}	3.87×10^{-1}
Clastic dike	5,400	6.96×10^{-5}	6.42×10^{-2}
Vadose zone 3 m thicker	5,500	6.51×10^{-5}	6.86×10^{-2}
U $K_d = 0$	5,400	2.09×10^{-4}	1.29×10^{-1}
$K_d = 0$ for all radionuclides	5,400	3.68×10^{-2}	8.69
Facility K_d's (concrete)	5,400	5.79×10^{-5}	3.72×10^{-2}
Capillary barrier (4.2 mm/year) Best Estimate Case	>13,000	1.71×10^{-10}	1.31×10^{-6}
Capillary barrier (50 mm/year)	9,300	7.33×10^{-9}	3.76×10^{-6}
Vertical hydraulic barrier	>13,000	0.00	7.51×10^{-7}
Shorter capillary barrier	>13,000	3.12×10^{-9}	4.41×10^{-6}
Capillary barrier failure	6,800	1.95×10^{-8}	1.45×10^{-2}
Sand backfill	1,800	1.64×10^{-3}	2.35x10-1
No side-slope effect	5,600	4.01×10^{-6}	4.48×10^{-2}
Forward rate	4,600	7.59×10^{-4}	6.54×10^{-1}
No ion exchange	5,500	6.13×10^{-5}	5.71×10^{-2}
No secondary phase formation	5,600	3.86×10^{-5}	3.57×10^{-2}
Include steel in waste packages	6,900	7.95×10^{-5}	6.98×10^{-2}
Top conditioning layer	5,000	7.72×10^{-5}	6.66×10^{-2}
Waste form aqueous diffusion (x10)	1,700	5.66×10^{-10}	1.61×10^{-3}
Alternative glass formulation (HLP31)	3,000	9.41×10^{-3}	6.23
Forward Rate + Ion Exchange Increased 5X	4,350	7.97×10^{-4}	7.46×10^{-1}

[a] Performance objective is 25 mrem/year
[b] Time of compliance is 1,000 years
[c] Estimated from base analysis case

Figure 4-35. Total Inventory-Normalized Contaminant Flux to the Groundwater as a Function of Recharge Rate (No Capillary Barrier).

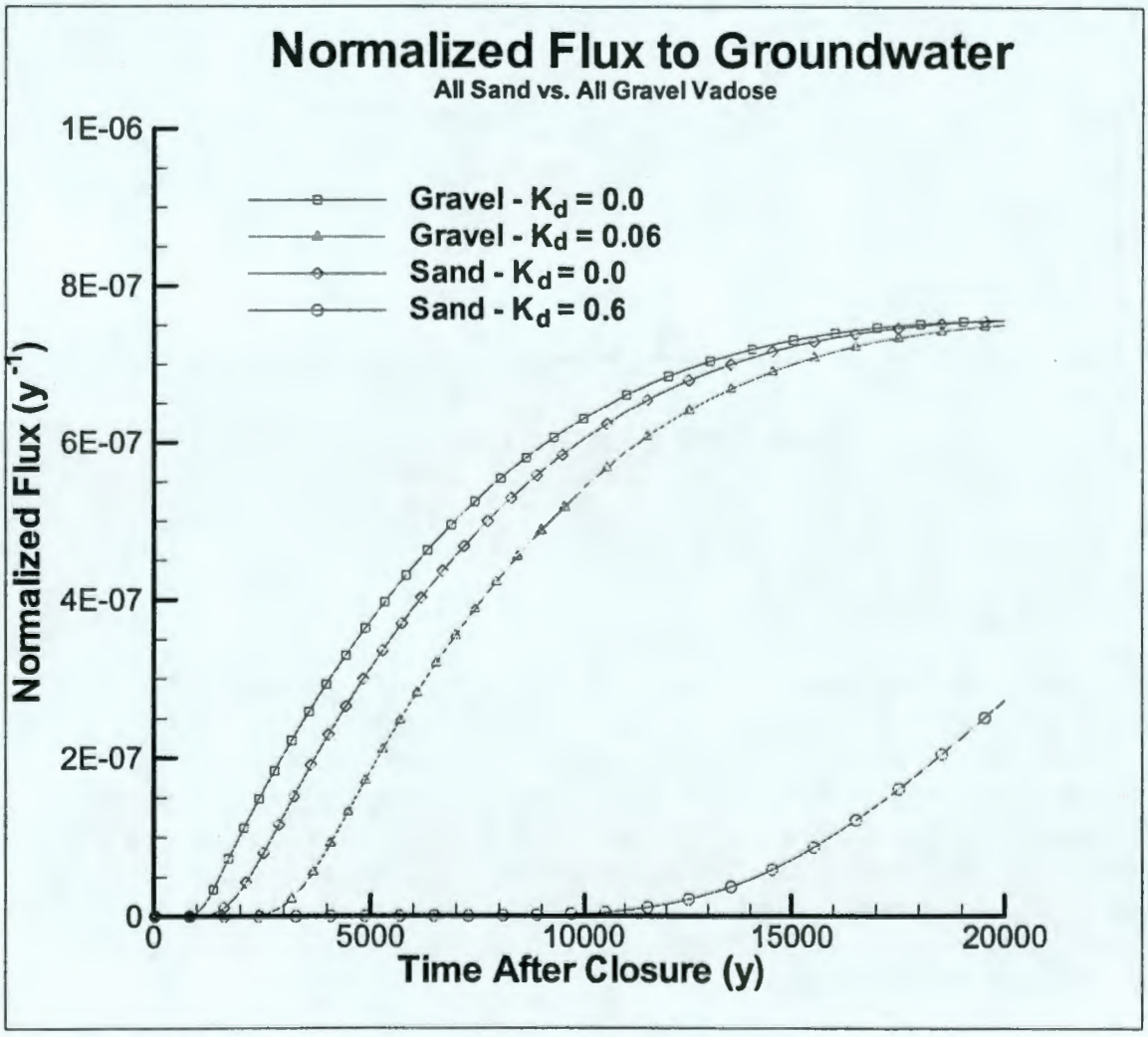


4.6.3 Vadose Zone Properties

The major vadose zone properties are the hydrology and geology parameters and the geochemical properties. The hydrology and geology parameters affecting the contaminant transport in the vadose zone are the hydraulic properties of the media, dispersivity and diffusivity. To explore the sensitivity to hydraulic properties, two sensitivity cases were run using the following assumptions: Assumed the vadose zone was all Hanford sand, and assumed the vadose zone was all Hanford gravels. Sensitivity cases were run exploring the sensitivity of the estimated impacts to the media's dispersivity and to diffusion of the contaminants: Increased the dispersivity by a factor of two and an isotropic calculation. Finally, a sensitivity case was run with the inclusion of a clastic dike.

Figure 4-36 shows the impact of assuming the entire vadose zone has hydraulic properties of either all sand or all gravel. Little impact occurs to the mobile contaminant flux to the aquifer as a function of time. However, the impact to less mobile contaminants is significant. This is caused by the gravel correction that is applied to the K_d's. Because of the limited surface area of gravels compared to sands or silts, the K_d for gravels is reduced. For the all-gravel case, the contaminants with $K_d = 0.6$ mL/g are significantly more mobile than in the base analysis case because the effective K_d for gravel is 0.06 mL/g for the entire vadose zone. Therefore, the travel time is faster. On the other hand, the all-sand case has a $K_d = 0.6$ mL/g assigned to the entire vadose zone and the corresponding travel times for these contaminants are significantly slower. The result is that at 10,000 years the alpha concentration is 2.54×10^{-3} pCi/L for the all-sand case, 3.39×10^{-2} pCi/L for the base analysis case, and 3.60×10^{-1}pCi/L for the all-gravel case. The corresponding all-pathways dose at 10,000 years is estimated to be 3.91×10^{-2} mrem/year for the all-sand case, 7.00×10^{-2} mrem/year for the base analysis case, and 3.87×10^{-1} mrem/year for the all-gravel case.

Figure 4-36. Total Inventory-Normalized Contaminant Flux to the Groundwater Assuming the Vadose Zone is All Sand or All Gravel (No Capillary Barrier - Recharge = 4.2 mm/year).



Changing the degree of dispersion and diffusion in the vadose zone affects the estimated impacts slightly. The travel times for the mobile contaminants are affected only slightly and the corresponding estimated beta-photon drinking water doses are not affected at all (see Appendix D, Table D-1). The less mobile contaminants are affected more significantly and the resulting all-pathways doses reflect these impacts. Specifically, an isotropic flow field in the vadose zone leads to lower estimated impacts for the all-pathways dose (see Table 4-13). Increased diffusion is anticipated to decrease transport. Dispersion tends to spread the contaminant plume laterally. For the base case, this moves the plume into the higher flow field associated with the side slope (recharge = 50 mm/year). Therefore, the estimated impacts should be higher with an increase in dispersion.

The geochemical properties affect the travel time for the contaminants through the vadose zone. Sensitivity cases were run that included the geochemical properties of a potential concrete layer beneath the remote handled waste disposal facility along with the geochemical properties of the vadose zone. The sensitivity case for the concrete layer assumed that the 1 m-thick concrete mixture layer had the chemical properties of degraded concrete (see Table 3-6 for probable values). The iodine and uranium contaminant transport is retarded in this region. The resulting estimated impacts to the all-pathway dose at 1,000 years after facility closure was 5.9×10^{-5} mrem/year (base case estimated impact is 7.8×10^{-5} mrem/year). The reduction in the estimated dose at 1,000 years was caused by a decrease in the contribution from ^{129}I. The resulting estimated impacts to the all-pathway dose at 10,000 years after facility closure was 0.037 mrem/year (base case estimated impact is 0.070 mrem/year). The reduction in the estimated dose at 10,000 years was primarily caused by a decrease in the uranium flux to the aquifer when compared to the base analysis case.

Changing geochemical parameters in the vadose zone does not change the travel time for mobile contaminants, but such a change determines which contaminants are mobile. The most important slightly retarded isotopes are the uranium isotopes. Changing the K_d value of uranium from 0.6 to 0 mL/g, increases the concentration of alpha emitters by approximately13 orders of magnitude to 0.00061 pCi/L at 1,000 years after facility closure (still far below the performance objective of 15. pCi/L). Decreasing the uranium K_d to 0 mL/g also increases the all-pathways dose by a factor of 3 at 1,000 years and by less than 2 at 10,000 years compared to the base analysis case.

If the retardation factors for all radioisotopes are set to zero, the estimated impacts increase even more. The estimated impacts are compared to the base analysis case in Table 4-14. At 1,000 years the estimated all-pathways dose is 3.68×10^{-2} mrem/year compared to the base analysis case estimate of 7.78×10^{-5} mrem/year. At 10,000 years the estimated all-pathways dose is 8.69 compared to 6.86×10^{-2} mrem/year for the base analysis case. Assigning $K_d = 0$ mL/g to the radium radionuclides leads to a estimated radium concentration of 5.95×10^{-3} pCi/L at 10,000 years compared to an estimate of 0 pCi/L for the base analysis case.

Table 4-14. Estimated Impacts from Sensitivity Case Where All Radionuclides are Assumed Mobile (K_d = 0 mL/g) Compared to the Base Analysis Case for a Well 100 m Downgradient from the Disposal Facility.

Type of Impact	Estimated Impacts (all K_d = 0 mL/g)		Estimated Impacts Base Analysis Case	
	1000 years*	10,000 years	1000 years*	10,000 years
Dose (mrem in a year) from beta-photon radionuclides in drinking water	0.000031	0.015	0.000021	0.0102
All-pathways dose (mrem in a year)	0.037	8.7	0.000078	0.070
Concentration (pCi/L) of alpha-emitting radionuclides	0.016	4.34	1.0×10^{-16}	0.034
Radium concentration (pCi/L)	0.0000038	0.0060	0.0	0.0
Uranium concentration (pCi/L)	0.00061	0.30	0.0	0.025

*Time of compliance is 1,000 years.

The final sensitivity case considered for the vadose zone is that of a clastic dike located beneath the disposal facility. The specific calculation assumes that a vertical clastic dike, 1 m wide, extends from a depth of -16 m to -99 m beneath each trench at a location 15 m from the trench centerline (see Finfrock [2000b] for details.) The remaining parameters are similar to the base analysis case. This presumed preferential flow path does not affect the contaminants reaching the aquifer. The reason for the relatively small effect for this preferred path is associated with the clastic dike acting as a capillary break with respect to the relatively high recharge from the side slope region (recharge = 50 mm/year). In comparing the results of the clastic dike estimated impacts to the case where the side slope recharge is 4.2 mm/year, the capillary break estimated impacts are higher as anticipated. These impacts are summarized in Table 4-15 and Appendix D, Table D-1.

Table 4-15. Estimated Impacts from Clastic Dike Simulation Compared to the Base Analysis Case for a Well 100 m Downgradient from the Disposal Facility.

Type of Impact	Estimated Impacts Clastic Dike		Estimated Impacts Base Analysis Case	
	1000 years*	10,000 years	1000 years*	10,000 years
Dose (mrem in a year) from beta-photon radionuclides in drinking water	0.000019	0.0101	0.000021	0.0102
All-pathways dose (mrem in a year)	0.000070	0.0642	0.000078	0.070
Concentration (pCi/L) of alpha-emitting radionuclides	9.16×10^{-17}	0.0281	1.0×10^{-16}	0.034

*Time of compliance is 1,000 years.

4.6.4 Waste Form Release or Disposal Facility Time Dependence

For most vadose zone calculations, the time dependence of the source release has no or at most a minor impact on the results of the vadose zone calculations. For these calculations the time dependence of the source is critical as seen from Figures 4-9 and 4-10, where the flux entering the groundwater has the same temporal shape as the flux leaving the disposal facility. Therefore, for each different waste form release or disposal facility case, a vadose zone calculation was run. These calculations show that the vadose zone's main influence is to delay the contaminants from reaching the groundwater. Thus, the details of the calculations are not presented here. Explicit cases were run for the following:

- Forward rate of waste form release
- No ion exchange in waste release
- Alternative glass composition
- Uranium release in near field
- Capillary barrier (best estimate case)
- Shorter barrier
- Barrier subsidence
- Vault as the disposal facility.

The results from these analyses are summarized in Appendix D, Table D-1.

4.7 SENSITIVITY CASES: GROUNDWATER FLOW AND CONTAMINANT TRANSPORT

4.7.1 Overview

Because of the short travel time in the groundwater compared to the waste form release rates and the vadose zone travel times, the groundwater calculations were separated from the other calculations. The results of the groundwater calculations are given in terms of the well intercept factor, which is the ratio of the inventory-normalized contaminant groundwater concentration at the point of interest (e.g. at the well head 100 meters downgradient from the disposal facility – expressed as Ci/m^3) to the total inventory-normalized contaminant concentration entering the groundwater (expressed as Ci/m^3).

Calculations were performed for the following variables:

- Disposal facility layouts and location
- Pumping rates at the well retrieving the contaminated water
- Hydraulic conductivities beneath the waste disposal site
- Regional influences.

Table 4-16 displays the ratios of the well intercept factors for the various groundwater sensitivity cases to the WIF of the base analysis case at two locations. Because the amount of water entering through the vadose zone directly under the disposal facility is insignificant compared to the total groundwater flow, impacts are proportional to the well intercept factor.

Table 4-16. Ratio of Well Intercept Factors for the Various Groundwater Sensitivity Cases.

Section	Description	WIF/(Base Case WIF)	
		Ratio at 100 m	Ratio at 1,000 m
4.3.5	Base case	--	--
4.7.2	Trenches at south end of site	1.50	1.16
4.7.2	90° rotation	1.91	1.36
4.7.2	Smaller layout	0.27	0.27
4.72	Larger layout	1.20	1.18
4.72	Existing vault site	0.43	0.07
4.7.3	30 L/day pumping	1.00	1.00
4.7.3	100 L/day pumping	1.00	1.00
4.7.3	300 L/day pumping	1.00	1.00
4.7.3	1000 L/day pumping	1.00	1.00
4.7.4	Reduced hydraulic conductivity by a factor of 3	11.98	4.86
4.7.5	Regional increase in recharge by a factor of 3	0.92	0.41
4.7.5	Regional decrease in recharge by a factor of 3	0.68	0.65
4.7.5	Decrease in regional upgradient boundaries by a factor of 2	0.84	0.82

4.7.2 Disposal Facility Layouts and Location

The layout of the disposal units at the disposal site has not been determined. In addition, the existing concrete vaults at the eastern edge of the 200 Area still might be used. The current planning is for the units to be placed in the north end of the site (the base analysis case). A series of different sensitivity cases were used to examine the effect of different assumptions on facility location and layout.

One sensitivity study examined the effect of locating the six disposal-facility trenches evaluated in the base case at the south end of the ILAW disposal-facility area. One of the key factors in the calculated WIF for the base-case analysis was the assumed hydrogeologic unit and corresponding hydraulic conductivity found at the water table directly below the facility. With the disposal trenches located in the northern part of the ILAW disposal facility area, the disposal facility is largely underlain by relatively high permeability sediments associated with the Hanford formation. Moving the disposal trenches to the south end of the facility area will position the disposal facility closer to the water-table contact between the Hanford formation and the lower permeability sediments associated with the Ringold Formation. The change in postulated hydraulic properties at the water table will result in a different velocity distribution beneath the facility that could affect calculated WIFs.

The direction of plume movement in this case is very similar to the base case, but calculated WIFs are 50 percent higher than for the base case at 100 m and about 16 percent higher at 1 km. This result is consistent with the postulation that, with a thinner distribution of Hanford formation sediments in the south end of the facility, the overall distribution of groundwater velocities would be lower and the resulting WIF would be higher than for the base case.

Another sensitivity case examined the effect on the WIF of rotating the orientation of the six remote handled trenches evaluated in the base case by 90 degrees. Conceptually, flow across the facility is predominantly in a northwest to southeast direction. The change in orientation would put the longest dimension of the individual remote handled trenches in an orientation closer to parallel to the dominant direction of flow. This would conceivably decrease the effective width of the disposal facility relative to groundwater flow and increase the magnitude of the WIF.

While changing the trench configuration did have some effect on the calculated WIFs, the resulting WIF at the 100 m (328-ft) well was 90 percent higher than the 100 m (328-ft) well WIFs calculated for the base case. The calculated WIF at 1 km (0.62 mi) was increased by about 36 percent over the 1 km (0.62 mi) WIF in the base case.

As seen in Figure 4-8, the contaminant flux does not have a significant horizontal component except in cases of very low recharge and vadose zone moisture content. The concrete vaults have a smaller footprint (36,750 m^2) than the trenches (124,800 m^2). Therefore, a case was run assuming the footprint of the concrete trenches. Differences between the WIFs calculated for this case and the remote-handled trench case are directly attributable to assumptions used for source release areas in both cases. The ratio of the WIFs between this case and the base case at 100 m are on the order of 0.27, which is reflective, though slightly lower, than the ratio of the release areas (0.29).

Other sensitivity cases examined the effect of increasing the effective surface area of release at the water table beyond the basic footprint of the base case. After transport through the vadose zone, contaminants originating from the individual disposal units would disperse in a pattern likely to be larger than the original footprint of the individual trench configuration. In this sensitivity case, two subcases were evaluated. One case evaluated a source-release area for the remote-handled trench concept reflective not only of the individual remote-handled trench areas, but of the intervening inter-trench areas. Another evaluated a source-release area for the concrete-vault concept reflective not only of the individual concrete vault areas, but of the intervening inter-trench areas.

Calculations for this case showed that the assumed 20 percent increase in the source-release area resulted in about a 20-percent increase in the WIFs over the base-case values at both the 100-m and 1-km wells. This result is consistent with the additional contaminant mass introduced at the water table for this case. This result, combined with previous results for remote-handled trench base case and the concrete vault releases, suggests a linear relationship between the source-release area and calculated WIFs over the range of the assumed release area.

Groundwater flow and transport also were calculated for the site located at the existing grout facility containing the four existing empty concrete vaults. The large differences between the WIFs calculated for this case and those calculated for the remote-handled trench case are attributable to assumptions used for source release areas and the lower estimated values for hydraulic properties used for hydrogeologic units in the existing grout facility model. The remote-handled trench calculations were based on an assumed release area of 124,800 m^2, reflecting the footprint of the assumed six-trench configuration. The concrete vault calculations were based on the assumed 2,560 m^2 release area, reflecting the footprint of the smaller four-concrete-vault configuration. The ratio of the WIFs between the two cases at 100 meters are on the order of 0.43, which is much higher than expected given that the ratio of the release area is on the order of 0.02. The higher than expected WIF in this case is caused by the lower hydraulic conductivities used in this local scale model. Hydraulic conductivities used for the Hanford formation beneath the existing grout facilities, which are on the order of 200 m/day to 300 m/day, are about 25 to 50 times lower than those beneath the new ILAW disposal facility area, which vary from about 6,500 m/day to 14,500 m/day). In general, the lower hydraulic conductivities used in the existing grout facility model contribute to lower pore water velocities and lower horizontal flow beneath the existing grout facility, and create an overall increase in the calculated WIF. The general increase in the WIF for this case reflects differences in the release area and the estimated hydraulic properties.

4.7.3 Pumping Rates at the Well Retrieving the Contaminated Water

Results of these sensitivity cases showed that pumping in the ranges of rates investigated would have little effect on the calculated WIFs. The effect of these relatively low pumping rates is consistent with the fact that water pumped at the 100 m (328-ft) well location is largely pulled from the Hanford formation soils. Given the magnitude of the estimated permeabilities of the Hanford formation at the location of the 100 m (328 ft) well (about 4,400 m/day), the hydraulic effect of the pumping would be minimal and would not significantly alter the local flow field and the overall plume movement. Calculated WIFs for these cases are virtually identical as those calculated at the 100 m (328 ft) well and at 1 km (0.62 mi) in the base case.

4.7.4 Hydraulic Conductivities Beneath the Site

The estimated hydraulic properties and interpretations of the distribution of major hydrogeologic units used in the Sitewide model and local-scale models are based on interpretations of limited measurements and well log information. Uncertainties in estimates of hydraulic properties and boundaries of the major units are associated with these interpretations. This sensitivity study investigates the effect of the position and the associated hydraulic-property differences between the Hanford formation and the underlying Ringold Formation (Unit 5). Directly beneath the disposal-facility area, the estimated hydraulic conductivities of the Hanford formation range from 2500 m/day to 30,000 m/day (27,340 to 32,808 yd/day). In contrast, the hydraulic conductivity of the Ringold Formation (Unit 5) ranges from 40 m/day to 350 m/day (44 to 383 yd/day). For this sensitivity study, the permeability of the Hanford formation where it exists beneath the disposal facility was lowered to the hydraulic-conductivity levels of the underlying Ringold Formation. Conceptually, this change effectively reduces simulated

velocities and flow rates in the hydrogeologic unit at the water table and would result in an increase in the calculated WIFs.

The reduction in hydraulic conductivities changed the primary direction of groundwater flow beneath the facility to a more easterly direction (Bergeron 2000). WIFs calculated at the 100-m (328-ft) well and at 1 km (0.62 mi) indicate that a reduction in the hydraulic conductivity of the underlying Hanford formation to those in the Ringold Formation (Unit 5) below the Hanford formation would increase calculated WIFs by about an order of magnitude (a factor of 12 higher) at the 100- m (328-ft) well. The resulting WIF for the 4.2 mm/year recharge rate at 1 km (0.62 mi) was calculated to be a factor of about 5 higher than at the same location in the base case.

4.7.5 Regional Influences

Results of earlier work by Lu (1996) and the results of this study have shown that the characteristics of the hydrogeologic unit and the estimated water table are an important consideration and will influence the calculated WIFs downgradient of the ILAW facility. The actual position of the water table in the far future is indeed uncertain, and a series of sensitivity studies were done to examine the effect of factors that could affect the position of the water table beneath the ILAW facility. The two main factors that could influence the water table position include the estimated levels of regional natural recharge and inflow onto the Hanford Site from upgradient offsite sources.

One sensitivity case examined the effect of increasing regional natural recharge on the regional and local water-table conditions. In this case, the recharge was increased by a factor of three in the Sitewide model, and the resulting predicted water table was used to evaluate the effect of these changes in the local-scale flow and transport model.

According to Bergeron (2000), the simulated change in natural recharge in the Sitewide model raised the regional water table and significantly changed the overall predicted regional flow path for the ILAW facility from southeast and east toward the Columbia River to a predominant flow path north through the gap between Gable Butte and Gable Mountain to the Columbia River. The discharge area to the Columbia River for these conditions is eventually in the vicinity of the 100 N Area.

Locally, the water table was raised by about 3 m (16 ft) in the vicinity of the new disposal facility, resulting in an increased saturation of the Hanford formation beneath the ILAW facility. Results for this case, summarized in Table 4-16, indicate about a 8-percent reduction in the calculated WIF over the base case WIF at the 100-m (328 ft) well location. At the 1-km (0.62 mi) location, the resultant WIF was 18 to 57 percent lower than the WIF at the 100-m well location in the base case for the same assumed recharge rate. The decrease in the WIF at both locations suggests an increased amount of dilution resulting from this case.

Another sensitivity case examined the effect of reducing regional natural recharge on the regional and local water table conditions. In this case, the recharge was reduced by a factor of three. Results of this simulated change in natural recharge in the Sitewide model lowered the regional water table, but did not significantly change the overall predicted regional flow path for

the ILAW facility from southeast and east toward the Columbia River. The discharge area into the Columbia River for these conditions is, as in the base case, in the vicinity of the old Hanford town site.

Locally, water-table conditions were changed slightly from the base conditions and the water table was lowered by about 1.2 m (4 ft) near the new disposal facility, resulting in a slight decrease in saturation of the Hanford formation beneath the ILAW facility. Although the water table dropped for this case, the overall simulated hydraulic gradient in the vicinity of the disposal facility was a factor of 1.9 higher than was calculated in the base case (1.25×10^{-4} m/m [meter/meter] versus 6.6×10^{-5} m/m). So although this case resulted in a decrease in the water table, the amount of water beneath the facility increased slightly resulting in an increase in dilution and an overall reduction in the calculated WIF for this case. Results for these simulated conditions indicate a 32-percent reduction in the calculated WIF over the base case at the 100- m (328-ft) well location (7.1×10^{-4} versus 1.05×10^{-3}) for the 4.2 mm/y recharge rate case. At the 1 km (0.62 mi) location, the resultant WIF (5.4×10^{-4}) was 35 percent lower than the WIF at the same location in the base case (7.8×10^{-4}) for the same assumed recharge rate.

A final sensitivity case examined the effect of reducing regional boundary fluxes on the regional and local water table conditions at the Cold Creek and Dry Creek entrances to the Hanford Site as well as recharge to the unconfined aquifer from springs emanating along the base of Rattlesnake Hills. In this case, the simulated boundary fluxes were reduced by a factor of two.

Results of the simulated change in natural recharge in the Sitewide model lowered the regional water table, but did not significantly change the overall predicted regional flow path for the ILAW facility from southeast and east toward the Columbia River. The discharge area into the Columbia River for these conditions is, as in the base case, in the vicinity of the old Hanford town site.

Locally, water-table conditions were changed slightly from the base conditions and the water table elevation was lowered by about 0.5 m (1.6 ft) near the new disposal facility resulting in a slight decrease in saturation of the Hanford formation and slight changes to flow conditions beneath the ILAW facility. As in the previous case, the water table dropped and the overall hydraulic gradient in the vicinity of the disposal facility was a factor of 1.5 higher than was calculated using the base case (1.0×10^{-4} m/m versus 6.6×10^{-5} m/m). As in Case 8, this case resulted in a decrease in the water table and the amount of water beneath the facility increased slightly, resulting in more dilution and an overall reduction in the calculated WIF for this case. Results for these simulated conditions indicate about a 16-percent reduction in the calculated WIF over the base case WIF at the 100 m (328-ft) well location (8.8×10^{-4} versus 1.05×10^{-3}) for the 4.2 mm/y recharge rate case. At the 1 km (0.62-mi) location, the resultant WIF (6.8×10^{-4}) was 18 percent lower than the WIF at the same location in the base case (7.8×10^{-4}) for the same assumed recharge rate.

4.8 SENSITIVITY CASES: OTHER FACTORS

The groundwater pathway results are very sensitive to ^{99}Tc inventory and well screen depth. Other modeling parameter choices have only minor effects.

4.8.1 Overview

This section describes the sensitivity to the amount of radionuclides in the waste (the inventory), the land use assumed, the library of dosimetry parameters chosen, the location of the facility, and calculational assumptions.

4.8.2 Inventory

As only a few radionuclides dominate the impacts, only a few sensitivity cases related to inventory are given. Table 4-17 displays the sensitivity of variables to key radioisotopes. Performing no technetium separation increases the beta-gamma drinking water and all-pathways dose by a factor of 3 (depending on the impact and the time period looked at). Reducing the amount of technetium to zero decreases the beta-gamma drinking water and all-pathways dose by a factor of between 2 and 3. Changing the amount of uranium has a major effect on the concentration of alpha emitters and a minor effect on the all-pathways dose at 10,000 years. Doubling the amount of iodine increases the beta-gamma drinking water dose by about 50% and the all-pathways dose by about 25%. Even if all the radioisotopes are at their bounding values, the estimated impacts only increase by a factor of 2.5 for the beta-photon drinking water and all-pathways doses, and by a factor of 5 for the alpha concentration (when compared to the base analysis case), still well below performance objectives.

Finally, the impacts were estimated assuming the upper bound estimates for all radionuclides and assuming technetium separations did not occur. These sets of calculations represent an extreme estimate for the total inventory in the waste. The results from these calculations increased the base analysis case estimated impacts by a factor of 5, still well below the performance objectives.

Chemicals from the ILAW contribute so little that even using the bounding inventory in the glass waste form makes little difference in the importance of the results (See Tables 4-8 and 4-9). Chemicals from the ILAW contribute so little that even using the bounding inventory makes little difference in the importance of the results (See Tables 4-8 and 4-9). A possible exception is the chromium in the 304L stainless steel container. Not only is there more chromium in the containers (~9×10^6 kg) than in the immobilized waste form (2.7×10^5 kg), but the assumed steel corrosion rate results in the release of its constituents faster than the glass waste form. Using the chromium release rates from Section 4.5.5.2 and assuming that all of the chromium is mobile, the estimated concentration in groundwater 100 meters down gradient at 1,000 years after disposal is 0.023 mg/L, peaking at 2,850 years at 1.13 mg/L, and falling to 0.070 at 10,000 years. The performance goal for chromium is only 0.05 mg/L. These calculations should be bounding as there are probably kinetic constraints on the rate of oxidation

Table 4-17. Sensitivity of Variables to Key Radioisotope Inventories.[a]

Case	Drinking Water				All-Pathways Dose (mrem/year)	
	βγ Dose (mrem/year)		α concentration (pCi/L)			
	1,000 y	10,000 y	1,000 y	10,000 y	1,000 y	10,000 y
Base analysis case	0.000021	0.0102	1.0×10^{-16}	0.034	0.000078	0.070
Increase Tc inventory to 5 times original amount (i.e., No Tc separation)	0.000070 (3.3)	0.034 (3.3)	1.0×10^{-16} (1.00)	0.034 (1.00)	0.00030 (3.9)	0.175 (2.5)
Reduce Tc to zero	0.000009 (0.43)	0.0043 (0.43)	1.0×10^{-16} (1.00)	0.034 (1.00)	0.000023 (0.30)	0.044 (0.63)
Double inventory of I	0.000030 (1.4)	0.015 (1.5)	1.0×10^{-16} (1.0)	0.034 (1.0)	0.00010 (1.3)	0.081 (1.2)
Double inventory of U	0.000021 (1.0)	0.0102 (1.0)	1.0×10^{-16} (1.0)	0.058 (1.7)	0.000078 (1.0)	0.076 (1.1)
All inventory at bounding values with Tc separation occurring	0.000055 (2.6)	0.027 (2.7)	4.8×10^{-16} (4.8)	0.19 (5.5)	0.00017 (2.2)	0.31 (4.4)
All inventory at bounding values and Tc inventory at maximum value	0.00010 (4.8)	0.049 (4.9)	4.8×10^{-15} (4.8)	0.19 (5.5)	0.00038 (4.9)	0.41 (5.9)

[a] The relative change compared to the base analysis case is given in parenthesis.

of Cr(III) to Cr(VI). In the present STORM simulations, global equilibrium with atmospheric levels of $O_2(g)$ was assumed, which results in the dominant oxidation state of Cr as +6 (Pourbaix, 1977). However, future simulations should consider kinetics in constraining the oxidation rate of Cr. Because Cr(III) is less soluble and less mobile than Cr(VI), the effect would be to dramatically lower projected Cr releases to the vadose zone.

4.8.3 Land Use

Land use could affect the disposal facility by changing the regional amount of water entering the aquifer or by the exposure pathways. The concept of changing the regional recharge rate is discussed in Section 4.7.5 and is relatively small. Changes in exposure pathways are discussed in Section 4.8.4.

4.8.4 Dosimetry Parameters

The base analysis case assumes that all knowledge of the disposal facility is lost and a residential farmer is exposed. Table 4-18 displays all-pathways doses for three other exposure scenarios (industrial, residential, and agricultural) for land located near the disposal facility. The industrial scenario represents potential doses to workers primarily caused by drinking well water. The resident scenario represents an individual living near the disposal site and being exposed to well water and produce grown in a garden irrigated with well water. The agricultural scenario represents an individual living on a farm and includes exposure to meat and dairy products from animals exposed to pastures irrigated with well water. As seen in Table 4.18, the estimated impact to the all-pathway dose at 1,000 years after facility closure range from 9.9×10^{-6} to 1.3×10^{-4} mrem/year with the industrial scenario estimated to have the lowest estimated dose and the agricultural scenario having the highest estimated dose. Similarly, at 10,000 years after facility closure, the all-pathway estimated doses range from 0.053 to 0.10 mrem/year, with the industrial scenario estimated to have the lowest estimated dose and the agricultural scenario having the highest estimated dose.

Table 4-18. All-Pathways Doses for Three Other Exposure Pathways for Points Located Near the Disposal Facility.

Case	All-Pathways Dose (mrem/y)	
	1,000 years*	10,000 years
Base analysis case (resident farmer)	0.000078	0.070
Industrial	0.0000099	0.053
Residential	0.000034	0.055
Agricultural	0.000126	0.100

* Time of Compliance is 1,000 years

A Native American scenario was investigated for locations near the Columbia River. Because the Native American scenario requires large amounts of water, such a scenario is likely only near the river. Use of this scenario increases the all-pathways dose from 7.3×10^{-6} to 2.8×10^{-5} mrem/year at 1,000 years after facility closure (neglecting travel time in the aquifer) when compared to the base analysis case. Similarly, the impact to the all-pathway dose at 10,000 years after facility closure is estimated to increase from 0.0065 to 0.022 mrem/year when compared to the base analysis case.

Finally, the sensitivity of the estimated impacts to a different set of dosimetry factors was estimated. Specifically, calculations were performed for the all-pathway farmer scenario using the DOE dose factors (see Rittmann 1999 and Appendix B). The resulting estimated impacts for the all-pathway dose at 1,000 and 10,000 years after facility closure were approximately 10 percent lower using the DOE dose factors.

4.8.5 Facility Location

The sensitivity to the location of the disposal facility is investigated in Section 4.7.2.

4.8.6 Ignored Processes and Conditions

The major processes involved in contaminant transport have been modeled. However, the following processes also should be discussed:

- Upward diffusion of contaminants
- Upward migration by capillary action
- Consumption of water in the glass dissolution process.

Upward diffusion and upward migration only become important with extremely low recharge rates. Both processes produce a characteristic time, T, which depends on the soil diffusivity, D, and on the length to be traversed, L,

$$T = L^2 / D. \tag{4.1}$$

For the case of upward diffusion, the equation can be derived from Fick's law (see Section 4.9). For the case of capillary rise, the equation can be derived by ignoring the gravity term in the flow equation. Using a length of 6 m (the total thickness of materials above the waste) and a soil diffusivity of 1.25×0^{-7} cm^2/second (from Section 3.4.3.2), the characteristic time is almost 100,000 years.

Moreover, as noted in Section 3.4.4.3, water must be present to dissolve the glass. Because the aquifer is so far below the disposal facility, capillary action could not bring water up to the waste. The only mechanism for providing the quantity of water needed to dissolve a significant amount of the waste form is recharge. Therefore, under extremely dry conditions, no contaminants would be mobile.

The current waste form calculations do not take into account water consumed by the corrosion process. From section 3.5.5.8 a recharge rate of 0.005 mm/year is estimated to be totally consumed by glass corrosion. This recharge rate is much smaller than the recharge rate for the base analysis case and the other sensitivity studies investigated in this performance assessment where the capillary break is ignored. However, the estimated recharge rates for the best estimate case are the order of 8.5×10^{-3} mm/year in the waste package region (see Table 4-11). Therefore, the estimated impacts for the sensitivity cases with a functioning capillary break are overestimated.

4.9 SENSITIVITY CASES: EXTREME CASES

4.9.1 Overview

In the review of the *Hanford Low-Level Tank Waste Interim Performance Assessment* (Mann 1996a), a visible stakeholder noted (Appendix F.1 of Mann 1998a) that the public is concerned about not only what is expected, but also how bad the situation could be if some assumption in the analysis is wrong. Such comments have carried important weight with the program. This chapter discusses extreme cases, which are considered to be impossible, at least for most of the waste, but provide insights into extreme bounds of system performance.

4.9.2 Instantaneous Waste Form Release

Sections 4.3.4 (base case) and 4.5 (sensitivity cases) describe the deterministic calculations used to estimate the contaminant release from the glass blocks. Such release is expected to be extremely small, taking hundreds of thousands of years to complete. An extreme case is that all of the waste is released at one time (or at least during a period that is much shorter than the vadose zone travel time). Although physically and chemically impossible, such a case demonstrates the importance of understanding the waste form release mechanism. This case also allows us to determine the impact of any potential contamination on the outside of the ILAW packages. Such contamination would be expected to release very quickly.

Figure 4-37 displays the contaminant flux entering the groundwater assuming that all contaminants release from the waste form at a constant rate over 1 year. The shape of such a flux follows a classic vadose zone transport curve. Table 4-19 displays the estimated impacts, assuming all contaminants are release over this short time period.

Figure 4-37. Inventory-Normalized Contaminant Flux Entering the Groundwater Assuming All Contaminants Release from the Waste Form at Once.

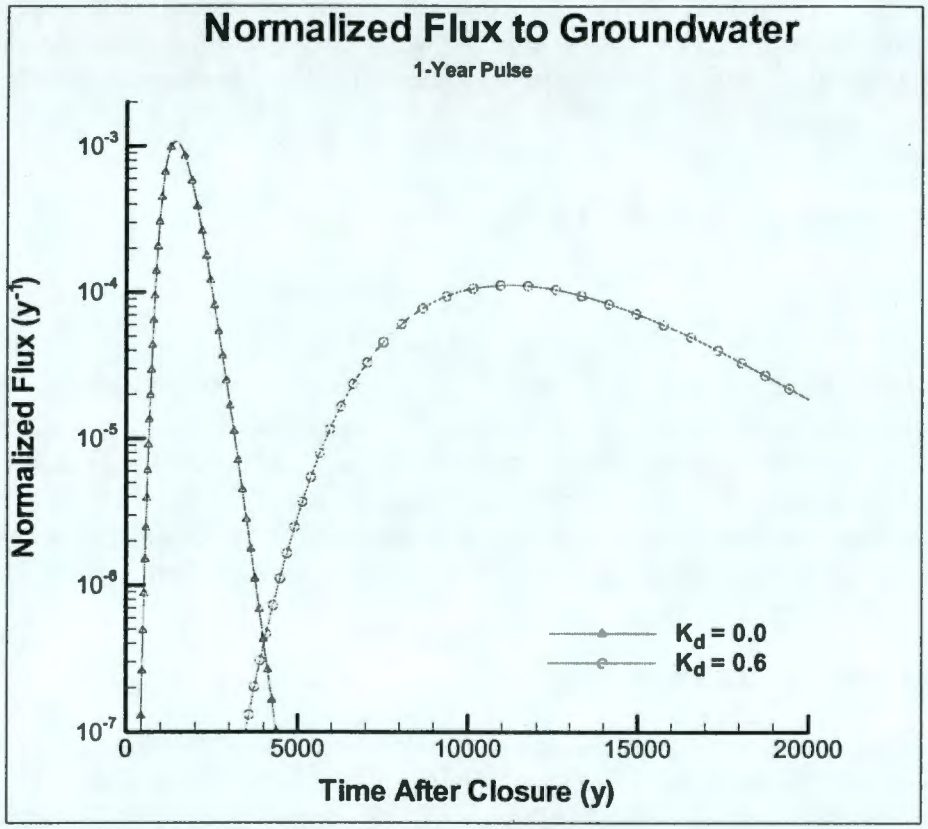


Table 4-19. Effect on Estimated Impacts Assuming All Contaminants Release at Once.

Impact	Base Analysis Case		Instantaneous Release		
	1,000 years	10,000 years	1,000 years	10,000 years	Maximum (Time of Peak)
βγ drinking water dose (mrem/y)	0.000021	0.0102	7.41	0.0	16.9 (1,385 y)
α concentration in water (pCi/L)	1.0×10^{-16}	0.034	3.6×10^{-11}	66.4	69.3 (11,150 y)
All-Pathways (mrem/year)	0.000078	0.070	27.1	62.9	67.2 (11,150 y)

There is about a five-order-of-magnitude increase for the impacts at 1,000 years. This reflects the importance of the slow release from glass. For the beta-gamma emitters, all of which are mobile, the contaminants travel through the vadose zone and the groundwater to the 100 m-

downgradient point-of-compliance well within 10,000 years. The peak impact for the beta-gamma drinking water dose is seen at 1,350 years. The alpha-emitters are retarded and are still rising at 10,000 years, reaching a peak at about 11,000 years. At 10,000 years, the all-pathways dose and the concentration of alpha emitters in the groundwater for the instantaneous release case is about two to three orders larger than the base analysis case. The beta/photon dose is insignificant at 10,000 years because the most important radioisotopes (i.e., those having $K_d = 0$) have already flowed past the point of compliance (see Figure 4-37).

However, for all impacts, even if there were an instantaneous release of the entire ILAW inventory, the impacts would be within a factor of five of performance objectives. Such low impacts from instantaneous release can be explained by the dispersion in the vadose zone and the dilution in the groundwater. By the time the contaminants reach the groundwater, travel in the vadose zone has increased the temporal spread to 980 years (the full width at the height that corresponds to half the maximum value). Such a spread is consistent with previous calculations for the ILAW site, as well as with other Hanford Site calculations for this travel time. The dilution in the groundwater is the same for this case as for the base analysis case.

The major contributors to the estimated impact to the beta-photon drinking water dose, the all-pathway dose, and the alpha concentration are summarized in Tables 4-20, 4-21, and 4-22, respectively.

Table 4-20. Major Radionuclide Contributors at 1,000 Years and 10,000 Years to the Estimated Beta-Photon Drinking Water Dose at a Well 100 m Downgradient from the Disposal Facility for Instantaneous Release.

Radionuclide	1,000 years*		10,000 years		Peak Exposure**	
	Dose (mrem/y)	Fraction	Dose (mrem/y)	Fraction	Dose (mrem/y)	Fraction
^{99}Tc	4.27	0.58	0.0	---	9.80	0.58
^{129}I	3.08	0.41	0.0	---	7.07	0.42
Other	0.05	0.01	0.0	---	0.03	----
Total	7.41	---	0.0	---	16.9	---

* Time of compliance is 1,000 years.
** See Table 4-19 for time of peak exposure.

Table 4-21. Major Radionuclide Contributors at 1,000 Years and 10,000 Years to the Estimated All-Pathways Dose at a Well 100 m Downgradient from the Disposal Facility for Instantaneous Release.

Radionuclide	1,000 years*		10,000 years		Peak Exposure**	
	Dose (mrem/y)	Fraction	Dose (mrem/y)	Fraction	Dose (mrem/y)	Fraction
^{99}Tc	19.2	0.71	0.0	0	0.0	0
^{129}I	7.8	0.29	0.0	0	0.0	0
^{231}Pa	---	---	0.93	0.02	1.05	0.02
^{233}U	---	---	5.95	0.09	6.34	0.09
^{234}U	---	---	1.99	0.03	2.13	0.03
^{238}U	---	---	2.06	0.03	2.20	0.03
^{237}Np	---	---	51.9	0.83	55.4	0.83
Other	0.1	0	0.07	0	0.08	0
Total	27.1	---	62.9	---	67.2	---

* Time of compliance is 1,000 years.
** For time of peak exposure, see Table 4-19.

Table 4-22. Major Radionuclide Contributors at 10,000 Years to the Estimated Concentration of Alpha Emitters at a Well 100 m Downgradient from the Disposal Facility for Instantaneous Release.

Radio-nuclide	1,000 years		10,000 years		Peak Exposure**	
	Concentration (pCi/L)	Fraction	Concentration (pCi/L)	Fraction	Concentration (pCi/L)	Fraction
^{231}Pa	0.0	---	0.13	0	0.15	0
^{233}U	0.0	---	27.16	0.42	28.95	0.42
^{234}U	0.0	---	9.32	0.14	9.95	0.14
^{238}U	0.0	---	10.17	0.16	10.85	0.16
^{235}U	0.0	---	0.38	0.01	0.41	0.01
^{236}U	0.0	---	0.32	0	0.34	0
^{237}Np	0.0	---	17.46	0.27	18.64	0.27
Other	0.0	---	0.01	0	0.01	0
Total	0.0	---	64.95	---	69.3	---

* Time of compliance is 1,000 years.
** For time of peak exposure, see Table 4-19.

4.9.3 Instantaneous Vadose Zone Travel

Sections 4.3.5 (base case) and 4.6 (sensitivity cases) describe the deterministic calculations of contaminant transport in the vadose zone. These calculations show that travel times are thousands of years for mobile contaminants to hundreds of thousands of years for more retarded species. An extreme case is to assume that the travel time is zero. Although contrary to our basic knowledge of moisture flow and contaminant movement, such a case shows the importance of the vadose zone for the performance assessment.

Table 4-23 displays the estimated impacts assuming that travel through the vadose zone is instantaneous. Release of waste glass contaminants directly to the groundwater has the following effects on the estimated doses when compared to the base analysis case. First, the beta-photon drinking water doses are significantly higher at 1,000 years after facility closure when compared to the base case. At 10,000 years after closure, the estimated impacts of an instantaneous transport through the vadose zone are comparable to the base case. This is because the waste glass release of contaminants is relatively constant after an initial transient period. Note that the peak beta-photon dose (0.80 mrem/year) occurs approximately 70 years after plant closure. The major contributors to this dose are ^{90}Sr (0.74 mrem/year) and ^{137}Cs (0.055 mrem/year). This highlights the important effect of the vadose zone to delay transport of radionuclides and allow those isotopes with relatively short half-lives to decay before reaching the aquifer. As shown in Table 4-23, the alpha concentration in the groundwater is much higher than the base case at both 1,000 and 10,000 years after facility closure. This result is caused by the additional retardation of contaminant flow for the alpha emitters in the base analysis case (see Figure 4-9).

For this sensitivity case, both the beta-photon drinking water and all-pathway dose performance objectives are met. The alpha concentration performance objectives also are met at 1,000 and 10,000 years after facility closure. For this extreme case, the radium radionuclides would reach the groundwater. At 1,000 years after facility closure, the radium concentration in the groundwater would be 3.5×10^{-4} pCi/L. At 10,000 years after facility closure, the radium concentration in the groundwater would be 5.8×10^{-3} pCi/L. Both these estimated impacts are well below the performance objective of 5 pCi/L. The total alpha concentration at 10,000 years after facility closure is 4.6 pCi/L, well below the performance objective of 15 pCi/L.

Table 4-24 compares the effect on the all-pathways dose when instantaneous transport through the vadose zone occurs for the most important radioisotopes. For the all-pathways dose, ^{239}Pu and ^{241}Am contribute over 89 percent of the estimated dose at 1,000 years after facility closure. At 10,000 years, ^{209}Po (2 percent), ^{229}Th (3 percent), ^{237}Np (4 percent), ^{239}Pu (82 percent), and ^{240}Pu (7 percent) contribute over 98 percent of the estimated dose. Except for ^{237}Np, all these radionuclides have assigned K_d = 150 mL/g and would not be expected to reach the aquifer in 10,000 years, even if the recharge rate were as high as 50 mm/year if the vadose zone was present.

Table 4-23. Estimated Impacts Assuming Instantaneous Vadose Zone Travel.

Impact	Base Analysis Case		Instantaneous Vadose Zone Travel		
	1,000 years	10,000 years	1,000 years	10,000 years	Maximum (Time of Peak)
βγ drinking water dose (mrem/y)	0.000021	0.0101	0.0029	0.016	0.80 (70 y)
α concentration in water (pCi/L)	1.0×10^{-16}	0.032	1.48	4.58	4.8 (13,100 y)
All-Pathways (mrem/y)	0.000078	0.069	3.39	9.36	9.4 (11,100 y)

Table 4-24. Effect On The All-Pathways Dose When Instantaneous Vadose Zone Travel Occurs For The Most Important Radioisotopes.

Radioisotope	Base Analysis Case		Instantaneous Vadose Zone Travel	
	1,000 years	10,000 years	1,000 years	10,000 years
^{99}Tc	0.0000552	0.0262	0.0051	0.028
^{129}I	0.0000224	0.0110	0.0021	0.012
^{231}Pa	0.0	0.000463	0.0007	0.006
^{233}U	0.0	0.00296	0.0071	0.040
^{234}U	0.0	0.000994	0.0024	0.014
^{238}U	0.0	0.00103	0.0024	0.014
^{237}Np	0.0	0.0258	0.0612	0.351
^{209}Po	----	----	0.0051	0.206
^{229}Th	----	----	0.0090	0.310
^{239}Pu	----	----	1.744	7.681
^{240}Pu	----	----	0.276	0.612
^{241}Am	----	----	1.268	0.000005
Other	0.0000002	0.0002	0.008	0.086
Total	0.0000778	0.0686	3.39	9.36

4.9.4 Bathtub Effect

The effect of trapping water within the trench for a period of time, then having a sudden release of the trapped water was investigated in Finfrock (2000b). A simplified calculation was performed assuming all free space within the trench was filled with water. A waste form calculation was performed to determine the amount of contaminant released from the waste form as a function of time (see Figure 4-32 and Bacon 2001). The released contaminant concentration then was assumed to instantly enter the groundwater. The peak all-pathway dose for this bounding calculation was estimated to be 2.3 mrem/year if the release occurred after 100 years and 8.1 mrem/year if the release occurred after 1,000 years. The major contributors to the all-pathway dose for the 100-year release case are ^{241}Am (41 percent), ^{239}Pu (14 percent), ^{90}Sr (36 percent), ^{137}Cs (6 percent), and ^{240}Pu (2 percent). The major contributors to the all-pathway dose for the 1,000-year release case are ^{241}Am (37 percent), ^{239}Pu (52 percent), ^{237}Np (2 percent), and ^{240}Pu (8 percent). If chemical adsorption is considered (i.e., if only those radionuclides with

K_d less than 4 mL/g are assumed to reach the groundwater with concentrations equivalent to the estimated concentrations in the trench before the release), then the peak all-pathway dose was estimated to be 0.015 mrem/year if the release occurred after 100 years and 0.19 mrem/year if the release occurred after 1,000 years. The major contributors to the all-pathway dose for the 100-year release case are ^{237}Np (73 percent), ^{99}Tc (6 percent), ^{129}I (3 percent), ^{233}U (9 percent), ^{238}U (3 percent), and ^{234}U (3 percent). The major contributors to the all-pathway dose for the 1,000-year release case are ^{237}Np (75 percent), ^{99}Tc (6 percent), ^{129}I (3 percent), ^{233}U (9 percent), ^{238}U (3 percent), and ^{234}U (3 percent). These are upper bound estimates because any spreading in the plume as it is transported through the vadose zone has not been included in the estimate. (Note that for the pulse case (i.e., the instantaneous release rate case discussed in Section 4.9.2), the peak concentration is reduced by a factor of 1,000 as the contaminants are transported through the vadose zone. The reduction is anticipated to be less than the pulse case.) Calculations performed in the 1998 ILAW PA (Mann 1998a) estimated this effect to impact the estimated dose by less than 3 percent if the effect were to last for 2000 years.

4.10 CUMULATIVE IMPACTS FROM HANFORD SITE ACTIVITIES

Other actions at the Hanford Site will have a minimal effect on this disposal action. This disposal action will have minimal effect on other Hanford Site actions.

4.10.1 Introduction

The Defense Nuclear Facilities Safety Board recommended (DNFSB 1994) that, when evaluating of the impact of the disposal of radioactive waste, DOE consider not just the impact from the disposal action under consideration but all other government actions that might affect those impacts. This recommendation has been inserted into DOE O 435.1 (DOE 1999b). The Richland Operations Office has submitted a composite analysis (Kincaid 1998) that addressed all the Hanford Site's disposal and closure actions on the Hanford Site's central plateau for which inventories had been established. The DOE has conditionally approved this composite analysis (DOE 1999d). The analysis presented here will focus on the results of the composite analysis and the results of the 1998 ILAW PA (Mann 1998a).

4.10.2 Hanford Site Composite Analysis

A Composite Analysis (Kincaid 1998) was submitted to DOE-Headquarters supporting the active and planned low-level waste disposal on the 200 Area Plateau. DOE (DOE 1999d) conditionally approved this composite analysis. It is a companion analysis to PAs for active and planned disposal, and remedial investigations and feasibility studies for remediation sites. With some limitations primarily based on available inventory data, this was a first iteration analysis to discover the potential long-term impacts to an offsite individual from all waste forms to reside at the Hanford Site at the time of Site closure. For the Composite Analysis, the offsite individual resided outside the exclusive waste management area and buffer zone as defined by the Hanford Future Site Uses Working Group (HFSUWG 1992). The analysis considered a 1,000 year regulatory period following Site closure and reached the following conclusions:

- Significant releases from the liquid discharge sites, tank leaks, losses from tanks during tank waste recovery operations, and pre-1988 solid waste burial grounds that occurred or began before Site closure were separated in time significantly from any discharges from the active or planned disposal.

- Peak all-pathways dose to the maximally exposed individual outside the buffer zone after Site closure (assumed in 2050) was less than 6 mrem in a year. This was the result of the agricultural scenario. Lesser doses resulted from the residential, industrial, and recreational scenarios. Despite high uncertainty in the inventories of past liquid discharges, tank leaks, tank losses, and pre-1988 burial grounds, this analyses indicated dose outside the buffer zone would not exceed the dose constraint of 30 mrem in a year.

- Maximum contaminant concentrations are highest now and will decline with time. Releases from liquid discharge sites, tank leaks, future tank losses, and pre-1988 burial grounds continue to enter the aquifer over the next few decades, but, in general, the rate of nuclide mass releasing to the aquifer will decline from now until Site closure and then continue to decline.

- No releases from the TWRS immobilized low-activity waste disposal facilities, the Environmental Restoration Disposal Facility trench and the graphite cores of the production reactors are seen during the regulatory period of 1,000 years or the full 1,500 years analyzed.

Future iterations of the Composite Analysis will be based on a more fully consistent inventory for all post-closure waste sites, including canyon buildings and associated facilities, which include permanent filters, and the PUREX tunnels.

4.10.3 Results from 1998 ILAW PA

Because the 1998 Hanford Site Composite Analysis (Kincaid 1998) performed simulations only out to 1,500 years and because this analysis goes out 10,000 years, this section provides an extended discussion of overlapping plumes. This section repeats the discussion that appeared in the 1998 ILAW PA (Mann 1998a).

One of the important differences in the current and past ILAW performance assessments relates to the groundwater models used in both assessments. Although no detailed comparisons between the current model and the model described in Lu (1996) have been made, some general observations about the two models and their past predictions were made in Cole (1997). In general, the results of past regional transport analyses of tritium, ^{129}I, ^{99}Tc, and uranium with the three-dimensional models described in Cole (1997) agreed with past Sitewide modeling results obtained by Chiaramonte (1996) in support of the Environmental Restoration Program. However, because of basic differences in assumptions made about the hydrogeologic framework and the horizontal and vertical discretization used in each model, discrepancies between model predictions on subregional and local-scales should be expected. A direct comparison of a calculated WIF for the Concept 1 source analyzed by Lu (1996) by Bergeron (2000) using the current model produced a WIF that was a factor of 10 lower than the WIF calculated by Lu

(1996) in the previous PA. The hydraulic conductivities used by the current model and the model used by Lu (1996) for the Ringold Formation are on the same order of magnitude, between 40 m/day and 300 m/day in the current model and between 70 m/day and 245 m/day in model used by Lu (1996). However, the current model contains areas of the Hanford formation beneath the facility and, as a result, has areas of very high permeability (between 2,200 m/day and 30,000 m/day) in the area of the source release.

The main pathway affected in an integrated analysis is the groundwater pathway. Inadvertent intruder scenarios and releases to the air are controlled at the source point. As will be seen in Section 4.14, the impacts of catastrophic events on waste that has been disposed are not significant.

Assuming that the disposal facility is placed just southwest of the PUREX facility, the previous Hanford Site actions that would affect groundwater downgradient of the disposal facility are those activities upgradient of the facility. From Figure 3-1 of the 1998 ILAW PA or from Figure 3-1 of this PA, those actions are the operations at the S, SX, SY, and U tank farms and the disposal of waste at the Environmental Remediation Disposal Facility and the commercial low-level waste disposal site operated by a commercial firm (U.S. Ecology).

Using the aquifer model described in Section 3.5.3.3 of the 1998 ILAW PA, mixing effects were calculated (appendix of Lu 1996) for the transport of contaminants from these potential sources to a well 100 m downgradient of the disposal facility. The mixing factors are displayed in Table 4-25. Because of the distance from the sources, the mixing factors are significantly less than from the low-level tank waste disposal site.

Table 4-25. Mixing Factors* from Other Sources for a Well 100 m Downgradient of the Low-Level Tank Waste Disposal Site.

	Upstream Sources				
	Low-Level Tank Waste Site	ERDF Site	S, SX, SY Tank Farm	U Tank Farm	Commercial LLW Disposal Site
Mixing Factor*	1.77×10^{-3}	3.01×10^{-4}	2.08×10^{-4}	1.25×10^{-4}	8.91×10^{-5}
Fraction of Base Analysis Case	1.00	0.17	0.12	0.07	0.05

*Mixing factor is the contaminant concentration in well water divided by the contamination concentration at the bottom of the vadose zone.

Both the DOE (DOE 1999b) and the NRC (10 CFR 61-4) mandate that a member of the general public can receive an exposure of no more than 25 mrem in a year at the edge of a disposal facility. Assuming that the other sources have their maximum impact and ignoring any time factors that might reduce the effect of the overlap, the other sources could add no more than 10.3 mrem in a year. Adding this to the amount estimated in Section 4.3.6 (all-pathway dose = 0.070 mrem/year) yields a total dose of less than 10.4 mrem in a year for the first 10,000 years. Moving the point of compliance to the 200 East Area fence will reduce the estimated compliance dose by less than 25 percent.

4.11 SUMMARY OF GROUNDWATER PATHWAY CASES

Estimated impacts to the groundwater resource have been calculated for the base analysis case. These estimated impacts are significantly lower than the performance objectives that have been established for this disposal action (see Section 1.6). A best estimate calculation has been performed for the proposed disposal action that indicates additional margin when compared to the base analysis case. Finally, a series of sensitivity calculations have been performed to provide the reader with an understanding of the relative sensitivity of these results to both hypothetical and extremely unlikely situations. The results taken in total provide a convincing argument that the proposed disposal of ILAW waste in a remote handled trench facility concept will not violate the performance objectives for protecting the groundwater resources.

The results from the base analysis case calculations are summarized in Tables 4-3 through 4-9. The base analysis case is based on site-specific data and includes conservative assumptions when needed to ease calculations. At 1,000 years after facility closure, the estimated impacts are more than 5 orders of magnitude less than the performance objectives for the radionuclides (Table 4-3). At 10,000 years after facility closure, the estimated impacts remain more than a factor of 400 less than the performance objectives. The estimated impacts for the chemicals associated with the glass waste are more than a factor of 1,000 less than the performance objectives (Table 4-8).

The results from the best estimate case calculations are summarized in Table 4-10. The best estimate case is based on a functioning capillary break located above the facility and on site-specific data, and includes conservative assumptions when needed to ease calculational resource demands. The lower moisture flux into the facility caused by the capillary break results in estimated impacts that are more than three orders of magnitude less than the estimated impacts for the base analysis case.

Sensitivity cases were calculated to investigate the sensitivity of the base analysis case to key parameters and assumptions. These key parameters and assumptions included moisture flow into the facility (Section 4.4), different facility concepts (Section 4.4), waste form performance (Section 4.5), vadose zone properties (Section 4.6), groundwater moisture flow assumptions (Section 4.7), inventory (Section 4.8.2), and dosimetry parameters (Section 4.8.4). In all cases the performance objectives for protecting the groundwater were met.

Extreme cases also were calculated to demonstrate worst case performance for the disposal action. The specific cases investigated included an instantaneous release of all contaminants from the waste form in 1 year, and the instantaneous transport of the contaminants through the vadose zone. Although these cases were highly incredible, the resulting impacts did not exceed the performance goals for protecting the groundwater by more than a factor of 10.

The calculations show that the peak values of impacts to groundwater are most sensitive to those parameters determining the inventory of ^{99}Tc, the rate of waste form release, and the amount of dilution in the groundwater. Recharge and geochemical parameters are important in establishing the time of arrival.

4.12 EFFECTS OF RELEASES TO AIR

This section describes the migration of nuclides from the glass waste, through the facility to the ground surface, and into the atmosphere. The primary transport mechanism is gaseous diffusion. Other mechanisms were considered and found to be insignificant. The initial inventory combined with the small glass dissolution rate and the delay associated with diffusion reduces the surface release rate to levels that are well below the performance objectives for the ILAW.

4.12.1 Upward Diffusion of Radionuclides

The principal mechanism by which nuclides migrate from the waste to the ground surface is gaseous diffusion. The diffusion of radioactive gases such as tritium (as water vapor), ^{14}C (as carbon dioxide), and ^{222}Rn (an inert gas) can be represented using Fick's law of diffusion with a loss term for radioactive decay (Jury 1991). The amount available for diffusion, i.e., the source concentration, is changing with time due to the corrosion of the glass and radioactive decay. Two cases must be considered because the ^{222}Rn is being produced by a decay chain that includes ^{238}U, ^{234}U, ^{230}Th, and ^{226}Ra.

For ^{3}H and ^{14}C the source concentration increases with time after closure due to corrosion of the glass and decreases with time due to radioactive decay. If we first assume that all of the ^{3}H and ^{14}C released during the year diffuses away from the trench, then the source concentration for these two radionuclides is shown below. This assumption will be tested by comparing the predicted total airborne emission rate with the estimated total corrosion rate of the glass. The two release rates should be comparable in a system near steady-state. The total amount entering the air each year after diffusion through the cover should not be greater than the total amount that is released from the glass each year.

$$C_0(t) = f\ C_G(0)\ e^{-\lambda t} \tag{4.2}$$

where

$C_0(t)$ = diffusion source concentration in the waste trench, i.e., the average concentration of gas that is free to diffuse through the cover soil, Ci/m^3

$C_G(0)$ = average gas concentration in the waste trench at closure (t=0), Ci/m^3. This is computed as the total inventory in the waste trench divided by the trench volume.

f = bounding annual waste dissolution fraction, 9.3×10^{-7} per y (Table 4-1). It is the fraction of the total that is released by corrosion of the glass.

t = elapsed time since closure, y

λ = radioactive decay constant for the nuclide under consideration, per y. It is computed as the natural logarithm of 2 divided by the half life in years.

The source concentration is largest initially. The calculation of the maximum source concentration is shown in Table 4-26.

Table 4-26. Calculation Summary for ^{3}H and ^{14}C

	^{3}H	^{14}C
Nuclide Half Life	12.33 y	5,730 y
Decay Constant, λ	0.05622 y^{-1}	1.210E-04 y^{-1}
Bounding Waste Inventory at Closure, A_G (from Table 3-1)	3,251 Ci	4,361 Ci
Average Trench Concentration at Closure, $C_G(0)$	4.59E-03 Ci/m^3	6.16E-03 Ci/m^3
Total Waste Release Rate at Closure, $f A_G$	3.02E-03 Ci/y	4.06E-03 Ci/y
Maximum Source Concentration, $C_0(0)$	4.27E-09 Ci/m^3	5.73E-09 Ci/m^3
Diffusion Flux into the Air, $J(z_0)$	2.68E-08 Ci/m^2 per y	3.62E-08 Ci/m^2 per y
Activity Airborne Annually	3.34E-03 Ci/y	4.51E-03 Ci/y
CAP88-PC Dose Factor	0.0237 mrem/Ci	1.32 mrem/Ci
Air Pathway Dose	7.91E-05 mrem/y	5.96E-03 mrem/y

Numbers for the radionuclide half life are from ENDF/B-VI.

Decay constants are the natural logarithm of 2 divided by the half life (1 y = 365.25 d).

The bounding inventories for ^{3}H and ^{14}C have been decayed 36 years corresponding to the difference between the inventory date (1994) and plant closure (2030).

The average concentration initially in the trenches is calculated as the bounding waste inventory at closure divided by the volume of the six waste disposal trenches, 708,000 m^3.

The total glass release rate is the initial inventory at closure times the bounding annual dissolution fraction, 9.3×10^{-7}.

The maximum source concentration is calculated from the trench concentration at closure, the bounding annual dissolution fraction, and the decay time since closure.

The diffusion flux into the air is calculated from the diffusion model described in the text.

The annual airborne emission from the disposal facility is calculated as the product of the bounding diffusion flux, the facility footprint area (124,800 m^2), and the release duration (1 y).

Unit release dose factors are from *Exposure Scenarios and Unit Dose Factors for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (Rittmann 1999).

For calculating diffusion in the presence of radioactive decay the source concentration is assumed to be constant because it changes very slowly with time. This assumption tends to exaggerate the diffusion flux at the surface. The steady-state diffusion equation is shown below. It also assumes the diffusion characteristics of the waste cover are uniform with depth.

$$D\frac{\partial^2 C}{\partial z^2} = \lambda C \tag{4.3}$$

The solution to the above equation has an exponential dependence in elevation as shown below. The boundary conditions that the soil concentration is C_0 at the waste ($z=0$) and zero at the surface ($z=z_0$) have been included. The solution is only valid from $z=0$ to $z=z_0$.

$$C = C_0\left[\frac{e^{-z\sqrt{\lambda/D}} - e^{-(2z_0 - z)\sqrt{\lambda/D}}}{1 - e^{-2z_0\sqrt{\lambda/D}}}\right] \tag{4.4}$$

$$J(z) = -D\frac{\partial C}{\partial z} \quad \text{thus} \quad J(z_0) = \frac{2\sqrt{\lambda D}\, C_0\, e^{-z_0\sqrt{\lambda/D}}}{1 - e^{-2z_0\sqrt{\lambda/D}}} \tag{4.5}$$

where

C = gas concentration at elevation z in the soil, Ci/m^3. At the bottom of the soil cover, the soil concentration matches the gas concentration in the waste, i.e., $C=C_0$. At the top of the cover the gas concentration is zero.

D = diffusion coefficient for low atomic number gases moving through soil, $0.01\ cm^2/s = 31.56\ m^2/y$

J = upward diffusion flux, Ci/m^2 per y

z = vertical position in the soil, m. The bottom of the soil column is $z=0$, while the ground surface is $z=z_0=5$ m.

λ = radioactive decay constant for the nuclide, per y.

Previous calculations did not use the boundary condition that the surface concentration is zero. In effect, an infinite medium was assumed. This underestimates the surface diffusion flux (J) by at least a factor of two. Longer half-life nuclides show a larger difference.

The diffusion flux at the surface is shown in Table 4-26. The small values for diffusion flux are largely due to the slow release rate from the glass. The values shown are bounding numbers for the following reasons.

- The estimated glass dissolution rate is 9.3×10^{-7} per year after 10,000 years. It is the peak glass dissolution rate for the base analysis case (see Section 4.3.3).

- Only ^{222}Rn is an inert gas that will faithfully follow the diffusion model. The ^{3}H and ^{14}C will undergo chemical reactions and be part of compounds that are likely to exist in the soil or the glass corrosion products. For ^{3}H these compounds include water and hydroxides. For ^{14}C these compounds include carbonates and carbides. The effect of these chemical reactions is to slow the migration to the surface and the resulting release rate from the ground surface. This effect has not been included in the diffusion calculations to maximize estimated consequences.

- The predicted airborne emission rate from the surface of the facility exceeds the estimated waste release rate. For ^{14}C, the total initial waste release rate due to corrosion is 0.00406 Ci/y while the predicted emission rate from the facility surface is 0.00451 Ci/y. The fact that the airborne emission rate is larger than the glass release rate suggests that the maximum source concentration should be smaller.

The source concentration of ^{222}Rn is based on the amount of ^{226}Ra that has dissolved. The ^{226}Ra produces ^{222}Rn by radioactive decay. The amount of ^{226}Ra slowly increases for two reasons. First, an increasing fraction of the glass matrix has dissolved. Second, ^{226}Ra is being produced by the radioactive decay of ^{238}U and ^{234}U. The peak ^{222}Rn flux occurs after all the glass has dissolved, and the ^{226}Ra has reached radioactive equilibrium with the ^{238}U. This equilibrium occurs after times greater than 1×10^{6} years after closure.

The initial inventory of ^{238}U in the disposal facility is 328 Ci. After a few million years, the glass matrix has released the uranium and the ^{230}Th, ^{226}Ra, and ^{222}Rn are all in radioactive equilibrium with the ^{238}U. Thus the ^{222}Rn inventory in the trench is 328 Ci, and the average concentration in the waste trench is 4.63×10^{-4} Ci/m^3. It has been assumed that no migration of the uranium out of the trench has taken place during this time.

The calculation of the diffusion flux for ^{222}Rn is carried out using the formula given previously. The diffusion flux at the ground surface is 3.03×10^{-5} Ci/m^2 per year, or 0.96 pCi/m^2 per second.

4.12.2 Mechanisms Affecting the Diffusion Flux

There are four environmental factors that can affect the rate of gas transport through the soil. These are barometric pressure, temperature, wind, and rain (Jury 1991). Each of these will be discussed in terms of its effect on the gaseous diffusion rate from the ground surface calculated above.

4.12.2.1 Atmospheric Pressure. An increase in atmospheric pressure will compress the air above the soil and drive air into the soil. The motion of air into the soil is impeded by the tortuosity of the flow channels and the small diameter of the surface openings between soil grains. When the atmospheric pressure decreases gases in the soil expand and the gas near the surface is released into the atmosphere. The effect on gases diffusing from underground is to slow the release rate while the atmospheric pressure increases, and increase the release rate when the atmospheric pressure decreases.

However, atmospheric pressure variation represents a very small change in the average pressure. The annual standard deviation of barometric pressure variations was at most 0.72% during the years 1988 through 1991 (Crippen 1993). The annual extremes for these years were at most 3.1% from the average. Such small variation suggests the influence of atmospheric pressure changes is small. This suspicion is born out in literature cited in *Soil Physics* (Jury 1991). A soil permeable column 3 m deep was affected by typical barometric pressure variations to a depth of at most 0.56 cm. Thus, barometric pressure changes will have little influence on the average diffusion flux from the surface.

4.12.2.2 Atmospheric Temperature. The air temperature at the ground surface varies sinusoidally during the day as well as during the year. The daily variation does not penetrate as deeply as the seasonal variation. The seasonal temperature change at a depth of 5 m in sandy soil is about 10% of the seasonal temperature change at the surface (Jury 1991). The variation in temperature decreases exponentially with distance below the surface. In addition, there is a phase shift that increases with depth. The cyclic variation in soil temperature will cause air movement due to the expansion and contraction of the air as the temperature increases and decreases. This air movement can be expected to increase the average diffusion flux at the surface.

However, the influx of air is limited to a shallow layer near the surface, just as with barometric pressure changes. The monthly average air temperature at Hanford typically ranges from -12° C to 30° C with an average around 12° C (Hoitink 1994). Thus the near-surface soil layer experiences a seasonal temperature variation no more than 10% on the absolute temperature scale. While this variation is an order of magnitude greater than the barometric pressure change, it is confined to a relatively shallow layer near the surface. Thus, temperature changes will have little effect on the average diffusion flux from the surface. If the 10% variation is assumed to accelerate gas transport in the top 10% of the soil cover, then the soil cover thickness is effectively reduced by 0.5 m. With a cover thickness of 4.5 m, the ^{222}Rn diffusion flux increases from 0.96 pCi/m^2 per second to 1.98 pCi/m^2 per second.

4.12.2.3 Wind. The motion of air over the ground surface will create areas with higher pressure and areas with lower pressure. This varying surface pressure is similar in effect to the

barometric pressure variation, but on a much shorter time scale. Experience with temperature changes suggests that the rapid variation will not penetrate as deeply into the soil. Thus the enhanced gaseous release rate will affect gas migration to a depth less than the 0.5 cm predicted for barometric pressure variation. It is concluded that wind effects will have little effect on the average diffusion flux from the surface.

4.12.2.4 Rainfall. As the water infiltrates the soil, it displaces some air from the soil pore space. As the water moves deeper, the pore space is refilled with air from the nearby pore space. This air motion will increase the mixing between adjacent layers and should increase the average diffusion flux as well.

However, as the water moves deeper in the soil, the water and air coexist in the pore space so that the motion of water has no effect on the diffusion of gases. Thus the rainfall effect is limited to a shallow surface layer. The overall effect on the average diffusion flux is very small. The largest recorded 24-h rainfall at the Hanford Meteorological Station is 4.85 cm (1.91 in), which occurred on October 1 and 2, 1957. Using a soil porosity of 35% means the water would saturate the soil to a depth of 14.0 cm (5.5 in). Thus the normal rainfall at Hanford affects gas motion in the soil to a depth considerably less than the normal temperature variation.

4.12.3 Consequences and Comparison with Performance Objectives

The only performance objective for diffusion flux applies to ^{222}Rn, and is 20 pCi/m^2 per second. The calculated bounding value for ^{222}Rn flux is 0.96 pCi/m^2 per second which is well below this objective.

The performance objectives for ^{3}H and ^{14}C are that the dose downwind be less than 10 mrem/y. The air pathway dose downwind was estimated using the formula shown below.

$$\text{Air Pathway Dose} = J\,A\,T\,DF \tag{4.6}$$

where

J = flux of the radionuclide from the soil surface, Ci/m^2 per y. Values are given in Table 4-26.

A = cross-sectional area of the disposal facility trenches when viewed from above, (6)(80 m)(260 m) = 124,800 m^2

T = duration of the release, 1 y

DF = air pathway dose factor from CAP88-PC for an annual release, mrem

The air transport factor and air pathway dose factors developed in Section 3.2.1 of *Exposure Scenarios and Unit Dose Factors for the Hanford Immobilized Low-Activity Tank*

Waste Performance Assessment (Rittmann 1999). The air transport factor is a bounding value that applies to annual emission near the border of a large area source. The dose factors were obtained using the CAP88-PC software from EPA (Parks 1992). The air pathway dose calculation is summarized in Table 4-26. The total dose from these air emissions (0.006 mrem/y) is well below the performance objective of 10 mrem/y. Note that the best estimate ILAW inventory in the waste form does not include either ^{3}H or ^{14}C.

4.13 EFFECTS FROM BIOTIC PATHWAYS

Because of the nature of the waste form (containerized glass logs) and the depth of placement of the waste placement (more than 5 m), biotic pathways are not credible. That is, the ability for plant and animal life to transport the contaminants is extremely unlikely. Neither plants nor animals have a direct means of access to waste over the time of compliance (1,000 years). Moreover, given the low moisture content inside the disposal facility (see Sections 4.3.2 and 4.3.7), plant roots would not survive.

4.14 EFFECTS OF CATASTROPHIC EVENTS

Based on the discussion presented in Section 3.3.2.4, the only scenario considered is that of an ice-age flood that scrapes away all material down to 20 m (the depth of the disposal facility), then redeposits the material over the area of the Hanford Site. As noted in Section 2.2.5.4, the next such event might be expected in about 50,000 years.

From the 1998 ILAW PA (Mann 1998a), using the farm scenario as the exposure scenario, and assuming that all the waste is in a form that can be inhaled or ingested, the predicted dose is estimated to be 0.76 mrem (EDE) in a year. Practically all the predicted dose (more than 98 percent) comes from the external exposure of ^{126}Sn and its daughters. If the glacier flood occurs as early as 10,000 years, the predicted dose increases to 1.01 mrem (EDE) in a year. If the flood occurs at 100,000 years, the predicted dose is 0.53 mrem (EDE) in a year. These values are much less than the 25 mrem (EDE) in a year maximum established for the first 10,000 years for the all-exposure pathways. If the flood only redistributes the waste over an area equivalent to the 200 Areas (78 km^2), the predicted dose at 50,000 years [14 mrem (EDE) in a year] is still less than the all-pathways limit. Because the current inventory estimates (Wootan 1999) are comparable to those used in the 1998 ILAW PA (with the ^{126}Sn inventory reduced by a factor of approximately 10), these estimated impacts are conservative.

4.15 ALARA ANALYSIS

> The Immobilized Waste Program is incorporating ALARA principles by the close cooperation of the performance assessment and design activities. When operations begin, they too will be integrated.

To keep exposures as low as reasonably achievable (ALARA) design, operations, and analysis projects must cooperate closely. The RPP Immobilized Waste Program is committed to such integration and already has had some success. The design of the waste treatment facility and disposal facility are expected to be optimized using the results of these studies.

4.15.1 Past ALARA Activities

Much of the early emphasis of the Immobilized Waste Program's performance assessment activity was to investigate the impact of various facility design options on long-term performance. These reports (Rawlins 1994 and Mann 1995d) helped guide early program thinking. The interim performance assessments (Mann 1996a and Mann 1997a) were used by the engineering staff in their alternatives generation analyses (for example, Burbank 1997) to investigate various disposal options, including the use of trenches and existing facilities as was discussed in Section 2.4.2.

The performance assessment activity was actively involved in selecting the new disposal site. Although many factors were considered (Shord 1995), the input from the performance assessment activity played a significant role.

The initial plans had ILAW disposal taking place in underground concrete vaults similar to those constructed for the grout disposal program. However, Burbank (1999) analyzed various options and found that the trench design had superior financial and scheduling qualities. At the time, the environmental impacts were thought similar. As documented in a preliminary analysis (Mann/Puigh 2000b) and confirmed by analyses in the document (Sections 4.4 and 4.5.6), in fact, ILAW disposal in a trench design has lower environmental impacts than the similar disposal in a concrete vault. The principles underlying this conclusion are that the chemical interactions between the disposal facility structures and the waste form are more important than the structural integrity of the disposal facility over the long times considered in this analysis. The life cycle cost savings are 272 million dollars (of 560 million dollars).

The specifications for waste form performance also have changed. In the initial request for proposals for waste treatment (DOE/RL 1996), the specifications were limits on fractional release rates for ^{99}Tc (2.8×10^{-14} second^{-1}) and for ^{79}Se, ^{129}I, ^{237}Np, and uranium isotopes (1.4×10^{-13} second^{-1}) based on a 7-day PCT (Product Consistency Test) measurement at 20 °C. Based on testing performed by the ILAW PA activity (McGrail 2001) and by EM-50 (Vienna 2000), it became obvious that this specification was incomplete. The present specification (DOE/ORP 2000c), which is heavily based on PA activities, states

"2.2.2.17 Waste Form Testing:

2.2.2.17.1 Leachability Index: The waste form shall have a sodium leachability index greater than 6.0 when tested for 90-days in deionized water using the ANSI/ANS-16.1 procedure.

2.2.2.17.2 Product Consistency Test: The normalized mass loss of sodium, silicon, and boron shall be measured using a seven-day product consistency test run at 90°C as defined in ASTM C1285-98. The test shall be conducted with a glass to water ratio of 1 gram of glass (-100 +200 mesh) per 10 milliliters of water. The normalized mass loss shall be less than 2.0 grams/m^2. Qualification testing shall include glass samples subjected to representative waste form cooling curves. The product consistency test shall be conducted on waste form samples that are statistically representative of the production glass.

2.2.2.17.3 Vapor Hydration Test: The glass corrosion rate shall be measured using a seven day vapor hydration test run at 200°C as defined in the DOE concurred upon Product and Secondary Waste Plan. The measured glass alteration rate shall be less than 50 grams/(m^2-day). Qualification testing shall include glass samples subjected to representative waste form cooling curves. The vapor hydration test shall be conducted on waste form samples that are statistically representative of the production glass."

4.15.2 Future ALARA Activities

The designs of both the waste treatment facility and the disposal facility have not been completed. Therefore, the sensitivity cases presented in this report may be used to aid the designers of these facilities in making choices that would minimize long-term impacts at a reasonable cost. The Office of River Protection and the River Protection Project are committed to using the results of this assessment to optimize designs.

Important information was gained in developing this performance assessment that can impact future ALARA analyses. The major information areas include the estimated impacts that inventory modifications, minimization of moisture movement through the disposal facility, contaminant release rates from the ILAW packages, chemical conditioning and trapping within the disposal facility and facility layout have on the environmental performance of the ILAW disposal system.

The inventory of key contaminants will depend not only on the amount of such contaminants presently stored in underground tanks, but also on the separation of such waste into low-activity and high-level waste streams. The design of the separations process can be optimized to minimize impacts from ILAW disposal while having minimal impacts on operational concerns and on the storage and disposal of the IHLW. Section 4.8.2 discusses the effect of varying the amount of technetium, iodine, and uranium in the ILAW packages and the resulting impact on groundwater impacts. Particularly in the case of technetium separation, this analysis provides useful information on the impact of including further technetium separation or even removing this separation process. Section 5.4.3.1 shows the impact on the intruder dose on changing key inventories. It should be noted that reduction of ^{137}Cs will have important near-term consequences; a fact that is driving the waste treatment contractor to remove as much ^{137}Cs as possible in order to meet their own ALARA concerns and drive down maintenance costs.

As shown in Section 4.5.4, reducing moisture flow through the disposal facility significantly reduces the impact on the contaminant release rate from the ILAW packages. In addition, Section 4.6.2 shows that the reduction of the moisture flow through the vadose zone greatly increases the travel time and hence reduces the impacts at the time of compliance. Thus, these sections show the potential value of various design features (surface covers [including the impact of varying dimensions], moisture diverters, vertical moisture barriers) that could such modify moisture flow as documented in Section 4.4. The potential inclusions of such features and their resulting cost benefit will be studied during the detailed design process.

The contaminant release rate from the ILAW packages is a major driver on the long-term estimated impacts. Extensive research programs are being performed by the waste treatment

contractor, by the Department of Energy's Environmental Management Science Program, as well as by the performance assessment activity. All of this information is being shared among these research programs. This combined knowledge will aid the waste treatment contractor in choosing the waste form composition that best balances the requirements of long-term performance, processibility, and costs. Already the long-term performance of the waste treatment contractor's glass has been improved by well over an order of magnitude. In addition, the results from Section 4.5, which show minimal impact from the composition of the container material, allow more freedom in the selection of container materials.

Special features can be placed in the facility to chemically change the environment. The disposal facility designers can use in the information from Section 4.5.5 to optimize the selection of backfill material and whether the presence of a layer, which would chemically condition the chemical properties of the incoming moisture, are cost effective. The results from Section 4.6.3 can be used to determine the effectiveness of a potential concrete layer to chemically trap iodine and uranium. The estimated impacts on the all-pathways dose from a 1 meter-thick concrete mixture at the bottom of the disposal trench are a 25% reduction at 1,000 years and a 50% reduction at 10,000 years.

The information in Section 4.7.2 can be used in the detail design of the layout of the disposal facility within the footprint of the disposal site. Placing the trenches at the southern end of the side would slightly increase all groundwater estimated impacts (by a factor of 50% at 100 meters and by 16% at 1,000 m) when compared to the groundwater estimated impacts if the trenches were at the northern end. Rotating the orientation of the trenches by 90 degrees would also increase the estimated groundwater pathway impacts.

The information presented above, along with other design data, will be used in the designs of both the waste treatment plant and the disposal facility. Studies have already begun on the appropriateness of technetium separation and the amount of waste loading in the ILAW packages. Similarly modifications to the conceptual design are being considered based on the results of this work. The results of such decisions will be documented in future performance assessments.

5.0 RESULTS FOR AN INADVERTENT INTRUDER SCENARIO

5.1 OVERVIEW

The DOE intends to control the disposal sites for as long as the waste may be dangerous to an intruder. However, DOE recognizes that an inadvertent intruder may happen onto the site and not be discovered until after exposure has occurred.

As noted in Section 1.6, the time of compliance is 500 years after closure. However, results will be presented for the period of 100 years to 1,000 years after closure. The performance objective for the driller scenario is 500 mrem (EDE) for a one-time exposure, while the performance objective for the homesteader scenario is 100 mrem (EDE) per year for a continuous exposure. The time of closure is taken to be 2030. All the analyses presented in this section assume that the surface barrier is in place (Section 2.4.1). All calculations were performed using a spreadsheet to calculate decay and to convert inventory concentrations into doses.

5.2 INADVERTENT INTRUDER SCENARIOS

The pathways described here assume that no memory of the disposal facility remains. The following two principal cases of intruders were considered:

- The disposal facility is compromised by irrigation for commercial farming. A large amount of water enters the disposal facility, causing increased contaminant release and increased transport of the contaminants to the unconfined aquifer.

- An inadvertent intruder digs or drills into the disposal site and brings some of the waste to the surface, receiving an acute dose. Another intruder (the homesteader) tills the waste into the soil and grows vegetables, receiving a continuous dose while engaged in various activities.

The scenario of irrigated farming on the disposal site is basically the same as the sensitivity case for the groundwater scenario where the infiltration rate is 50 mm/year. However, instead of a small amount of water naturally infiltrating the disposal facility, a much larger amount of water enters the disposal facility (see Section 4.6.2). The larger amount of water increases the contaminant release rate from the engineered facility. The increased amount of water certainly speeds the transport of the contaminants through the vadose zone into the unconfined aquifer. The estimated impacts are higher than for the base analysis case, but are still significantly lower than the performance objectives. If regional irrigation occurs, the well intercept factor from Section 4.7.2 is reduced by a factor of 0.92, indicating additional dilution. This scenario will not be addressed further in this section.

The following three intruder scenarios that involve bringing waste from the disposal facility to the surface usually are considered in a performance assessment.

- Excavating for a basement or building foundation
- Drilling for groundwater or minerals
- Living where waste has been exhumed and scattered over the surface.

Scenario 1 is not considered credible because the top of the waste is 5 m (16.4 ft) or more below the surface. Neither basements for home residences nor foundations for commercial structures are likely to extend this far below the surface. This scenario was not evaluated in these analyses.

Scenario 2, the construction of small water wells, is possible. The driller scenario begins with the assumption that some time after disposal operations have ended, a well is drilled through the waste. Drilling at the disposal site is unintentional, and the waste is not recognized as a potential hazard, even though it is assumed to be in the form of glass chunks. The waste, along with uncontaminated soil taken from the well, is spread over a work area near the well. The dose to the worker is the sum of the contributions from inhalation of resuspended dust, ingestion of trace amounts of soil, and external exposure at the center of a slab of contaminated soil.

Scenario 3 considers a family planting a garden using the material taken from the well. Each individual of concern receives dose by direct exposure to the radiation field in the garden, by inhaling resuspended dust, by ingesting trace amounts of soil, and by consuming garden produce. Given than a well is constructed (scenario 2), it is possible, if not probable, that some sort of homesteading will occur.

5.3 FACILITY DESCRIPTION AND ASSUMPTIONS FOR THE INADVERTENT INTRUDER ANALYSIS

Selecting values for parameters important in inadvertent intruder scenarios is difficult. Because such intrusion is postulated to be in the future, the nature of the intrusion is ill defined. Moreover, uncertainty abounds about the proper values to be used in a given scenario. DOE O 435.1 provides no specific guidance on the intruder scenario analysis. For this report the specific exposure scenario is defined in Rittmann (1999) and is based on intruder scenarios analyzed in earlier Hanford Site PAs (Wood 1995a, Wood 1996, and Mann 1998a).

For the inadvertent intruder scenarios, the most important variables are the amount of waste exhumed, the size of the area over which the waste is spread, and the physical integrity of the waste. Additional parameters, such as exposure time and inhalation rates, also are important but are not typically treated as variables.

The amount of waste material taken from the disposal site is assumed to be the maximum average waste distribution times the area of the borehole for the well. The maximum average waste distribution for the trench design is calculated for the central region for the trench (200 m x 20 m) and is defined as the waste volume divided by the area of this central region. For this performance assessment, the diameter of the well is assumed to be 0.3 m (1 ft). Although consistent with the diameters used in earlier Hanford Site performance assessments, this value is larger than the range of well diameters commonly found in local communities (10.2 cm to 25.4 cm [4 to 10 in.]). Table 5-1 provides the facility parameters used for the remote-handled waste trench and the alternative concept, the concrete vault.

The area over which the driller spreads the waste is assumed to be 100 m^2 (about 1,100 ft^2). This value has been historically used in Hanford Site performance assessments. The homesteader is assumed to spread the waste over a garden, which is taken to be 200 m^2. In earlier Hanford Site performance assessments, the garden area has been as large as 2,500 m^2 (0.62 acre). The 200 m^2 garden was chosen for this performance assessment because the size represents an area large enough to supply a significant portion of a person's vegetable and fruit diet and because the smaller size produces a higher dose, making it the conservative scenario. Household gardens in the vicinity of the Hanford Site range in size from 10 m^2 to 1,000 m^2 (107 ft^2 to 0.25 acre) (Napier 1984). In both scenarios the soil mixing depth is assumed to be 15 cm (5.9 in.). This value has been used in other onsite performance assessments and is the typical rooting depth for garden vegetables.

The integrity of the waste form becomes important in determining the amount of radionuclides available for inhalation or uptake by plants and animals. For the base case, 99 percent of the radionuclides exhumed are assumed to stay within the waste form (i.e., only 1 percent is available for ingestion and inhalation). This is slightly less than in previous studies, which considered only corrosion of the waste form, and accounts for some mechanical erosion of the waste form during drilling. The importance of this assumption was investigated by sensitivity cases.

The worker at the well drilling site is assumed to be exposed 8 hours a day for 5 days. The dose to the worker is the sum of the contributions from inhaling resuspended dust (0.12 mg/hour), ingesting trace amounts of soil (100 mg/day), and external exposure at the center of a slab of contaminated soil for 40 hours. The dose factors for this scenario can be found in Rittmann (1999) and are summarized in Appendix B, Table B-3. The homesteader is assumed to be exposed for 1 year. The soil inhalation rate for the homesteader is 573 mg/year. The incidental ingestion rate is the same as for the driller, 100 mg/day. The resulting dose factors are given in Rittmann (1999) and summarized in Appendix B, Table B-4.

Table 5-1. Facility Dimensions and Waste Volume Exhumed.

	Remote-Handled Waste Trench[1]	Concrete Vault
Waste package dimensions (m)[2]	1.4 x 1.4 x 1.2	1.4 x 1.4 x 1.2
Facility dimensions (m)	200.0 x 20.0	18.3 x 21.0
Layers of waste packages	4	6
Packages per layer[3]	1584	168
Surface area (m^2)	4000	384
Waste volume (m^3)	14900	2380
Average areal distribution of waste (m^3/m^2)	3.73	6.20
Volume waste exhumed (m^3)	0.272	0.453

[1]Only the central portion of the trench is considered. If the fringe regions were averaged in, the areal distribution and volume exhumed would be lower.
[2]The waste package is 1.4 m high but is filled only 85 percent full, so it is treated here as being 1.2 m high.
[3] The number of packages in the bottom layer in the trench is used as an approximation for the number of packages above the central region in the upper layers.

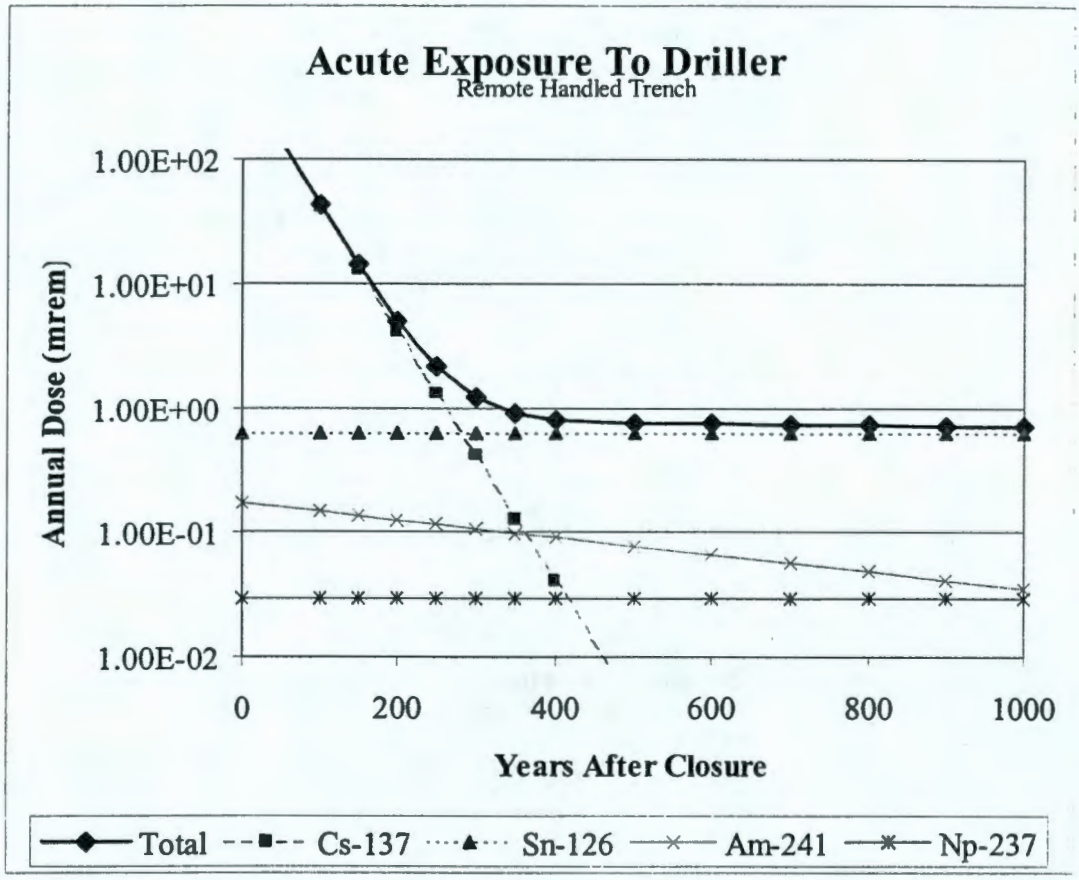
5.4 INADVERTENT INTRUDER ANALYSIS RESULTS

5.4.1 Driller Scenario

The results for the driller scenario are presented in Table 5-2 and displayed in Figure 5-1. The estimated acute exposure dose at 100 years after facility closure is 44 mrem. The major contributor to the acute dose is ^{137}Cs, which contributes approximately 95 percent of the exposure dose. ^{90}Sr and ^{126}Sn contribute 2.9 and 1.4 percent, respectively.

The estimated acute exposure dose at 500 years after facility closure (time of compliance) is 0.8 mrem. The major contributor to the acute dose is ^{126}Sn, which contributes approximately 82 percent of the dose. ^{241}Am and ^{237}Np contribute 10 and 4 percent, respectively. As discussed in Section 3.2, approximately 36 percent of the ^{126}Sn, 44 percent of the ^{237}Np, and 10 percent of the ^{241}Am of the initial tank inventories remain in the waste after processing (Kirkbride 1999). These numbers may change when specific flowsheet information is obtained.

Figure 5-1. Inadvertent Intruder Results for the Driller Scenario - Base Intruder Case.



These estimates are sensitive to the parameters assumed for the scenario. A number of sensitivity studies were performed to quantify the dependance on these parameters. The results of these studies are discussed in Section 5.4.3. None of the sensitivity cases changes the conclusion that the driller scenario meets the performance objective.

5.4.2 Homesteader Scenario

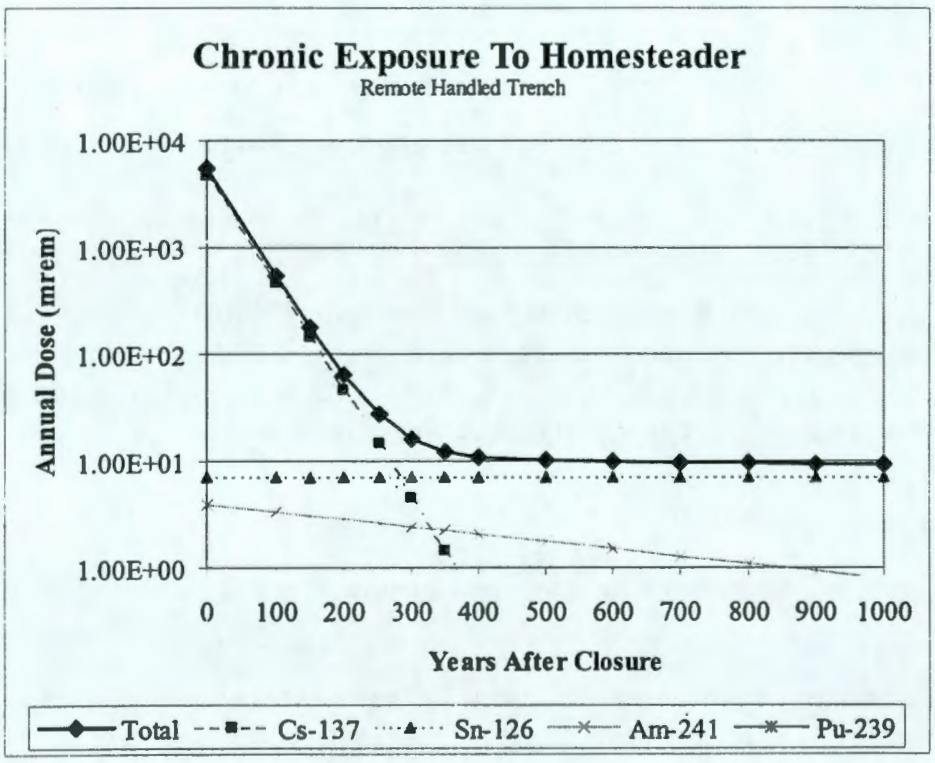
The results for the homesteader scenario are presented in Table 5-3 and displayed in Figure 5-2.

The estimated chronic exposure dose at 100 years after facility closure is 541 mrem. The dose remains above 100 mrem per year out to approximately 150 years. The major contributor to the acute dose is ^{137}Cs, which contributes approximately 85 percent of the dose. ^{90}Sr and ^{126}Sn contribute 13 and 1.3 percent, respectively. The estimated chronic exposure dose at 500 years after facility closure (time of compliance) is 10.2 mrem. The major contributor to the

homesteader dose is ^{126}Sn, which contributes approximately 69 percent of the dose. ^{241}Am and ^{239}Pu contribute 17 and 6 percent, respectively. As discussed in Section 3.2, approximately 36 percent of the ^{126}Sn, 5.5 percent of the ^{239}Pu, and 10 percent of the ^{241}Am of the initial tank inventories remain in the waste after processing (Kirkbride 1999). These numbers may change when specific flowsheet information is obtained.

These estimates are sensitive to the parameters assumed for the scenario. A number of sensitivity studies were performed to quantify the dependance of the estimated intruder impacts on these parameters. The results of these studies are discussed in Section 5.4.3. The only sensitivity case where the estimated impact exceeds the performance objective is the case that investigates having all the waste packages in the stack to have the maximum batch concentration of the key radionuclides (^{126}Sn and ^{241}Am). The remaining sensitivity cases do not change the conclusion that the homesteader scenario meets the performance objective.

Figure 5-2. Inadvertent Intruder Results for the Homesteader Scenario - Base Intruder Case.



5.4.3 Uncertainties

5.4.3.1 Waste Loading. To estimate the uncertainty in the inadvertent intruder estimated dose contributed by uncertainties in the inventory, the maximum batch concentrations listed in Table 3-1 were used. The maximum batch concentration reflects the tank-to-tank variation in inventory for each radionuclide. For example, the ratio of the maximum batch concentration to the average batch concentration for ^{126}Sn, ^{239}Pu, and ^{241}Am are 3.9, 4.9, and 25, respectively.

If just one of the waste packages affected by the driller had the maximum batch concentration and the remaining 3 packages (assuming a 4-layer stack) had average batch concentrations, the estimated acute exposure would be approximately a factor of 1.1 higher than the base case at 100 years and a factor 2.3 higher at 500 years. This would result in a dose that is still well below the performance objectives. For the homesteader scenario, the ratios would be 1.2 at 100 years and 2.7 at 500 years. This also would meet performance objectives.

If all of the packages in the facility (or all of the packages affected by the drilling) were maximum batch packages, the dose to the driller would be a factor of 1.5 higher than the base case at 100 years, and a factor of 6.1 higher than the base case at 500 years. This would result in a dose that is still well below the performance objectives. For the homesteader, the ratios would be 1.6 at 100 years and 7.6 at 500 years. In this case, the dose would still be below the performance objectives.

These estimated impacts can be mitigated through operational controls based on projected container inventories. Such operational controls will be better defined as the project matures.

5.4.3.2 Stability of Waste Form. It is assumed that the waste form will not completely shatter into fine grains. Rather, most of the radionuclides would remain in the waste form, making them unavailable for ingestion or inhalation. The base case for this analysis assumed 1 percent of the waste was available for ingestion or inhalation. For this case, the exposure is dominated by external radiation rather than by ingestion or inhalation. Previous ILAW performance assessments have assumed smaller numbers for the availability of material for ingestion or inhalation based only on corrosion (ignoring mechanical processes). Sensitivity studies were performed to assess the impact of assuming that 10 or 100 percent of the radionuclides are available for ingestion or inhalation. Increasing the availability of materials for ingestion or inhalation from 1 to 10 percent increased the dose to the driller by approximately 10 percent. The same change increased the dose to the homesteader by a factor of approximately 2. Increasing the availability from 1 percent to 100% increased the dose to driller by a factor of about 2, compared to a factor of 15 for the homesteader. The results of these analyses are shown in Figures 5-3 and 5-4.

Figure 5-3. Predicted Driller Doses for Different Availability of Radionuclides for Ingestion and Inhalation.

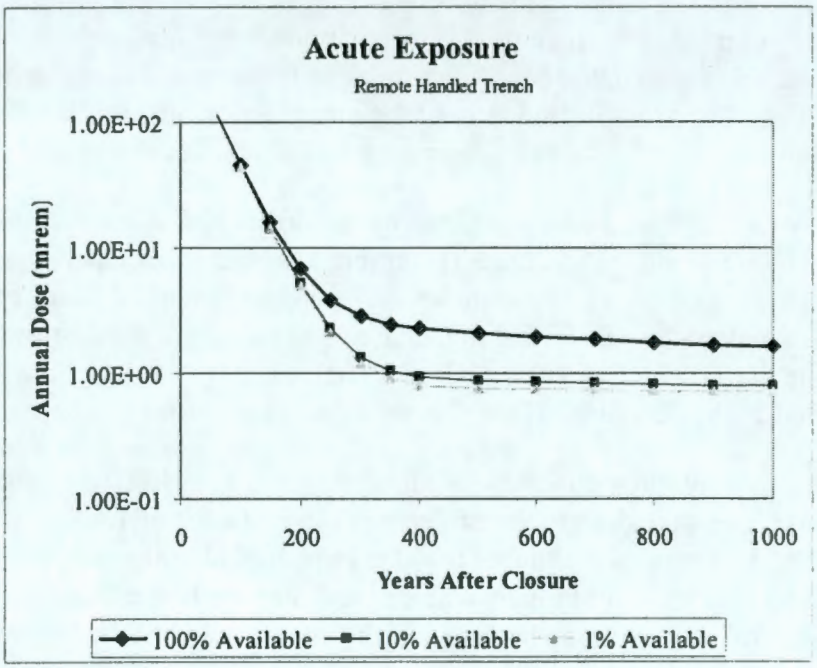
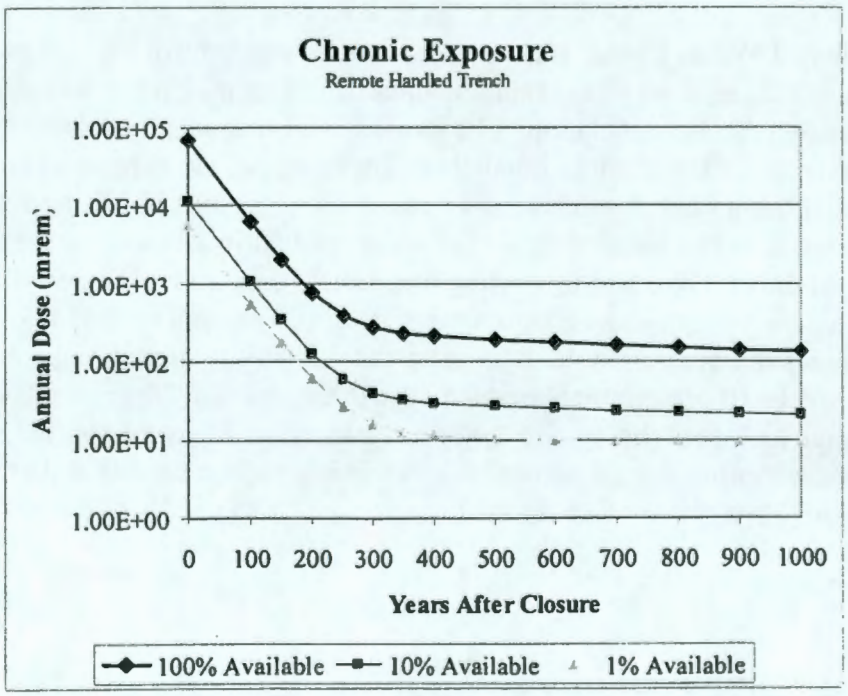


Figure 5-4. Predicted Homesteader Doses for Different Availability of Radionuclides for Ingestion and Inhalation.



5.4.3.3 Area of Homesteader Garden. The predicted dose depends on the area of the homesteader garden and the amount of time the homesteader spends in the garden. The homesteader receives dose from external radiation when working in the garden or dose from internal exposure when eating produce from the garden. The radionuclide concentration in the soil, and consequently the dose rate, is inversely proportional to the size of the garden, which implies that a smaller garden will produce a larger dose. This is partially offset by the assumption that the homesteader will spend more time in a larger garden (the exposure time will be proportional to the garden size).

The base intruder case assumes a much smaller area (200 m^2) than assumed in previous Hanford Site performance assessments (up to 2,500 m^2). The change in garden size causes the internal dose estimates to increase by a factor that is the inverse of the ratio of the garden sizes (up to 12.5). However, the external doses from the larger garden are only 10 to 25 percent less than the external dose from the smaller garden because of the shorter time spent in the smaller garden. Therefore, where direct doses dominate, the smaller garden area produces only a moderate increase in total dose. For radionuclides where ingestion is the main exposure pathway (^{90}Sr, ^{99}Tc, plutonium isotopes, ^{241}Am), the ratio of the predicted dose is inversely proportional to the ratio of the garden areas.

5.4.3.4 Size of the Hole Drilled. The predicted dose is proportional to the amount of waste brought up from the disposal facility. For the base intruder case, the hole was conservatively estimated as 0.3 m (12 in.). However, a more likely maximum diameter is 0.2 m (8 in). Such a smaller diameter hole would bring up only 42 percent of the waste that would come from the 0.3-m (12-in.) hole. This reduces the predicted dose by a factor of 0.42.

5.4.3.5 Average Areal Distribution of the Waste. Because the predicted dose depends linearly on the amount of waste material brought to the surface, the predicted dose depends on the waste concentration. The waste is assumed to be averaged over both the container and the filler material. However, because the hole is assumed to go through the entire height of the disposal facility, the vertical dimension becomes unimportant. This makes the average areal distribution (the volume of waste beneath each square meter of facility surface area [see Table 5-1]), rather than the normal density, important.

The average areal distribution of the disposal facility could differ from the base intruder case for any of the following:

- A different design of the disposal facility (for example, using dirt spacers between vault rows)
- Different ratios of waste material to filler material in a vault

- Different assumptions of the total height of waste
- Different waste loading (because of sodium content or other reasons).

The new contract (DOE/ORP 2000c) has altered the waste package designs to have a height of 2.3 m. These packages would be stacked 3 high. This would result in a stacking height of 6.9 m. This stacking height would increase doses by a factor of 1.44 compared to the base analysis case (assuming the package footprint, spacing, and waste loading remained constant).

The alternative (concrete vault) concept results in 0.453 m^3 of waste exhumed versus 0.272 m^3 (see Table 5-1). This corresponds to increasing the dose by a factor of 1.67.

5.4.3.6 Other Factors. The remaining factors are less important in determining the predicted dose. Such factors include the use of dose conversion factors and estimates of the amount of secondary radionuclides (i.e., those that don't dominate the dose) present.

The dose conversion factors important to the homesteader scenario are the external dose conversion factors as this exposure pathway dominates over ingestion or inhalation. The factors used in the base intruder case are the ones from the EPA (EPA 1993). For ^{126}Sn, the DOE external dose conversion factor (DOE 1988c) is a factor of 4.45 larger than the EPA factor. The DOE value is larger because it assumes all the radionuclides are on the surface of the soil and no shielding of the photon radiation occurs. The EPA factor, on the other hand, is based on the more realistic assumption of mixing over a 15 cm layer. Earlier Hanford Site PAs have used dose factors based on GENII (Napier 1988), which uses models similar to the EPA model. For ^{126}Sn, the external dose conversion factor from GENII is 20 percent higher than the EPA value. For ^{137}Cs, the ratios of the external dose conversion factors are similar to those for ^{126}Sn. For ^{90}Sr, ^{239}Pu, and ^{241}Am, the ratios of the GENII external dose conversion factors to the EPA values are 1.69, 1.11, and 0.66.

Because ^{126}Sn and ^{137}Cs dominate the intruder dose results, changes in the amount of other radionuclides is relatively unimportant. Using different processing strategies, the amount of ^{90}Sr might be as much as twice the amount assumed in the base intruder case (9.0 MCi versus 4.5 MCi). Even using the maximum amount of ^{90}Sr would increase the total predicted dose at 100 years by less than 15 percent because ^{90}Sr does not significantly contribute to the estimated dose unless it is ingested. Changes in the amount of other isotopes are predicted to be less significant to the intruder dose.

5.5 SUMMARY OF THE INADVERTENT INTRUDER SCENARIO

The estimated dose for the driller scenario is 44 mrem at 100 years and 0.8 mrem at 500 years. This meets the performance objective by more than two orders of magnitude. At the time of compliance the dose originates primarily from ^{126}Sn.

The estimated dose for the homesteader scenario is 541 mrem at 100 years and 10.2 mrem at 500 years. This easily meets the performance objective of 100 mrem at 500 years. At the time of compliance the dose originates primarily from ^{126}Sn.

Table 5-2. Driller Intruder Dose (mrem) at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
Co-60	3.38E-07	6.57E-13	1.28E-18	2.49E-24	4.85E-30	9.44E-36	3.58E-47	1.36E-58
Ni-59	8.11E-10	8.1E-10	8.09E-10	8.09E-10	8.08E-10	8.07E-10	8.06E-10	8.04E-10
Ni-63	7.15E-08	3.58E-08	1.79E-08	8.96E-09	4.48E-09	2.24E-09	5.62E-10	1.41E-10
Se-79	3.13E-07	3.13E-07	3.13E-07	3.13E-07	3.13E-07	3.13E-07	3.13E-07	3.13E-07
Sr-90+D	1.274391	0.108614	0.009257	0.000789	6.72E-05	5.73E-06	4.16E-08	3.02E-10
Zr-93	4.71E-05	4.72E-05	4.72E-05	4.72E-05	4.72E-05	4.72E-05	4.72E-05	4.72E-05
Nb-93m	8.83E-08	1.2E-09	1.63E-11	2.22E-13	3.03E-15	4.12E-17	7.62E-21	1.41E-24
Tc-99	0.000253	0.000253	0.000253	0.000253	0.000253	0.000253	0.000252	0.000252
Ru-106+D	2.05E-41	5.29E-71	1.4E-100	3.5E-130	9E-160	2.3E-189	1.5E-248	1E-307
Cd-113m	2.27E-06	1.67E-08	1.22E-10	8.95E-13	6.56E-15	4.81E-17	2.58E-21	1.39E-25
Sn-126+D	0.623751	0.623575	0.623399	0.623224	0.623048	0.622873	0.622522	0.622171
Sb-125	4.04E-14	3.8E-25	3.57E-36	3.35E-47	3.15E-58	2.96E-69	2.61E-91	2.3E-113
I-129	9.94E-05	9.94E-05	9.94E-05	9.94E-05	9.94E-05	9.94E-05	9.94E-05	9.94E-05
Cs-134	2.4E-18	6.03E-33	1.52E-47	3.81E-62	9.59E-77	2.41E-91	1.5E-120	9.6E-150
Cs-137+D	41.40264	4.107346	0.407469	0.040423	0.00401	0.000398	3.92E-06	3.85E-08
Sm-151	0.000124	5.74E-05	2.66E-05	1.23E-05	5.7E-06	2.64E-06	5.65E-07	1.21E-07
Eu-152	0.000546	3.01E-06	1.66E-08	9.17E-11	5.06E-13	2.8E-15	9.51E-18	9.43E-18
Eu-154	0.001486	4.66E-07	1.46E-10	4.58E-14	1.44E-17	4.51E-21	4.43E-28	4.36E-35
Eu-155	3.57E-09	1.32E-15	4.88E-22	1.81E-28	6.67E-35	2.47E-41	3.37E-54	4.6E-67
Ra-226+D	0.000177	0.00017	0.000162	0.000156	0.000149	0.000143	0.000131	0.00012
Ra-228+D	1.54E-08	8.94E-14	5.2E-19	3.02E-24	1.76E-29	1.02E-34	3.45E-45	1.17E-55
Ac-227+D	5.47E-07	2.26E-08	9.38E-10	3.88E-11	1.61E-12	6.66E-14	1.14E-16	1.96E-19
Th-229+D	0.000175	0.000173	0.000171	0.00017	0.000168	0.000167	0.000164	0.00016

Table 5-2. Driller Intruder Dose (mrem) at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
Th-232	0.006012	0.006012	0.006012	0.006012	0.006012	0.006012	0.006012	0.006012
Pa-231	0.000255	0.000257	0.000257	0.000256	0.000256	0.000255	0.000254	0.000253
U-232	0.026695	0.009889	0.003663	0.001357	0.000503	0.000186	2.56E-05	3.51E-06
U-233	0.000994	0.001624	0.002248	0.002866	0.003478	0.004084	0.005277	0.006448
U-234	3.29E-05	4.32E-05	5.88E-05	7.94E-05	0.000105	0.000135	0.000207	0.000296
U-235+D	0.000464	0.000467	0.000469	0.000472	0.000475	0.000478	0.000483	0.000489
U-236	7.61E-07	7.61E-07	7.61E-07	7.61E-07	7.61E-07	7.61E-07	7.61E-07	7.61E-07
U-238+D	0.001975	0.001975	0.001975	0.001975	0.001975	0.001975	0.001975	0.001975
Np-237+D	0.029629	0.029628	0.029628	0.029627	0.029626	0.029625	0.029623	0.029621
Pu-238	5.41E-05	2.46E-05	1.12E-05	5.07E-06	2.31E-06	1.06E-06	2.37E-07	6.76E-08
Pu-239	0.005087	0.005072	0.005058	0.005043	0.005029	0.005014	0.004986	0.004957
Pu-240	0.000841	0.000832	0.000824	0.000815	0.000806	0.000798	0.000781	0.000765
Pu-241+D	0.003338	0.002851	0.002432	0.002074	0.00177	0.00151	0.001101	0.000804
Pu-242	6.97E-08	6.97E-08	6.97E-08	6.97E-08	6.97E-08	6.97E-08	6.96E-08	6.96E-08
Am-241	0.147413	0.125711	0.107222	0.09147	0.078049	0.066614	0.048573	0.035477
Am-243+D	0.000208	0.000206	0.000204	0.000202	0.0002	0.000199	0.000195	0.000191
Cm-242	1.5E-07	6.8E-08	3.09E-08	1.4E-08	6.41E-09	2.94E-09	6.57E-10	1.87E-10
Cm-243	4.88E-05	4.32E-06	4.08E-07	6.47E-08	3.43E-08	3.14E-08	3.08E-08	3.04E-08
Cm-244	9.68E-07	4.52E-07	4.37E-07	4.32E-07	4.27E-07	4.23E-07	4.14E-07	4.05E-07
Total	43.52674	5.024938	1.200949	0.807429	0.756135	0.740875	0.722715	0.710143

Note: Nuclides with "+D" added to their name include the contributions from short-lived progeny, which are assumed to be in equilibrium at all times

Table 5-3. Homesteader Intruder Dose (mrem/year) at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
Co-60	3.55E-06	6.91E-12	1.35E-17	2.62E-23	5.1E-29	9.92E-35	3.76E-46	1.42E-57
Ni-59	1.35E-05	1.35E-05	1.35E-05	1.34E-05	1.34E-05	1.34E-05	1.34E-05	1.34E-05
Ni-63	0.001387	0.000694	0.000347	0.000174	8.69E-05	4.35E-05	1.09E-05	2.73E-06
Se-79	0.000105	0.000105	0.000105	0.000105	0.000105	0.000105	0.000105	0.000105
Sr-90+D	68.36017	5.82622	0.496559	0.042321	0.003607	0.000307	2.23E-06	1.62E-08
Zr-93	0.000729	0.00073	0.00073	0.00073	0.00073	0.00073	0.00073	0.00073
Nb-93m	1.08E-06	1.47E-08	1.99E-10	2.71E-12	3.69E-14	5.02E-16	9.29E-20	1.72E-23
Tc-99	0.080603	0.080576	0.08055	0.080524	0.080497	0.080471	0.080418	0.080365
Ru-106+D	1.67E-40	4.31E-70	1.1E-99	2.9E-129	7.3E-159	1.9E-188	1.3E-247	8.3E-307
Cd-113m	0.002179	1.6E-05	1.17E-07	8.58E-10	6.29E-12	4.61E-14	2.48E-18	1.33E-22
Sn-126+D	7.008526	7.006551	7.004577	7.002604	7.000631	6.998659	6.994716	6.990775
Sb-125	4E-13	3.76E-24	3.53E-35	3.32E-46	3.12E-57	2.93E-68	2.59E-90	2.3E-112
I-129	0.003508	0.003508	0.003508	0.003508	0.003508	0.003508	0.003508	0.003508
Cs-134	2.29E-17	5.77E-32	1.45E-46	3.64E-61	9.16E-76	2.3E-90	1.5E-119	9.2E-149
Cs-137+D	460.9885	45.73233	4.536872	0.45008	0.04465	0.00443	4.36E-05	4.29E-07
Sm-151	0.009765	0.00452	0.002093	0.000969	0.000448	0.000208	4.45E-05	9.53E-06
Eu-152	0.005985	3.3E-05	1.82E-07	1.01E-09	5.55E-12	3.13E-14	6.74E-16	6.73E-16
Eu-154	0.016049	5.03E-06	1.58E-09	4.95E-13	1.55E-16	4.87E-20	4.79E-27	4.71E-34
Eu-155	3.74E-08	1.38E-14	5.1E-21	1.89E-27	6.97E-34	2.58E-40	3.52E-53	4.81E-66
Ra-226+D	0.002021	0.001936	0.001854	0.001775	0.0017	0.001628	0.001493	0.001369
Ra-228+D	1.63E-07	9.5E-13	5.52E-18	3.21E-23	1.87E-28	1.09E-33	3.67E-44	1.24E-54
Ac-227+D	7.38E-06	3.06E-07	1.27E-08	5.24E-10	2.17E-11	8.99E-13	1.54E-15	2.65E-18

Table 5-3. Homesteader Intruder Dose (mrem/year) at Various Intrusion Times (years after closure).

Nuclide	100 y	200 y	300 y	400 y	500 y	600 y	800 y	1000 y
Th-229+D	0.002137	0.002117	0.002097	0.002077	0.002058	0.002038	0.002	0.001963
Th-232	0.068329	0.068329	0.068329	0.068329	0.068329	0.068329	0.068329	0.068329
Pa-231	0.003634	0.003666	0.00366	0.003652	0.003645	0.003637	0.003621	0.003606
U-232	0.300969	0.111491	0.041301	0.0153	0.005668	0.0021	0.000288	3.95E-05
U-233	0.017478	0.025185	0.032816	0.040371	0.047852	0.05526	0.069856	0.084166
U-234	0.002165	0.002286	0.002466	0.002704	0.002995	0.003339	0.004172	0.005188
U-235+D	0.005238	0.005278	0.005319	0.005359	0.005399	0.005439	0.005519	0.005599
U-236	6.36E-05	6.36E-05	6.36E-05	6.36E-05	6.35E-05	6.35E-05	6.35E-05	6.35E-05
U-238+D	0.023784	0.023784	0.023784	0.023784	0.023784	0.023784	0.023784	0.023784
Np-237+D	0.354485	0.354473	0.354462	0.35445	0.354439	0.354427	0.354404	0.354381
Pu-238	0.006677	0.00303	0.001376	0.000625	0.000285	0.00013	2.82E-05	7.24E-06
Pu-239	0.619496	0.617718	0.615945	0.614176	0.612413	0.610655	0.607154	0.603673
Pu-240	0.105175	0.10407	0.102977	0.101895	0.100825	0.099765	0.09768	0.095638
Pu-241+D	0.075626	0.064539	0.055018	0.046905	0.039993	0.034104	0.024813	0.018069
Pu-242	8.7E-06	8.7E-06	8.69E-06	8.69E-06	8.69E-06	8.69E-06	8.69E-06	8.68E-06
Am-241	3.338169	2.845473	2.425705	2.068071	1.763374	1.503777	1.094172	0.796851
Am-243+D	0.002472	0.002449	0.002427	0.002405	0.002382	0.002361	0.002317	0.002275
Cm-242	1.85E-05	8.39E-06	3.81E-06	1.73E-06	7.87E-07	3.59E-07	7.72E-08	1.91E-08
Cm-243	0.000576	5.23E-05	6.26E-06	2.21E-06	1.85E-06	1.81E-06	1.8E-06	1.78E-06
Cm-244	0.000119	5.65E-05	5.46E-05	5.4E-05	5.34E-05	5.29E-05	5.17E-05	5.07E-05
Total	**541.4061**	**62.89131**	**15.86502**	**10.93303**	**10.16954**	**9.859373**	**9.439349**	**9.140571**

Note: Nuclides with ""+D" added to their name include the contributions from short-lived progeny, which are assumed to be in equilibrium at all times

6.0 INTERPRETATION OF RESULTS

6.1 OVERVIEW

This chapter integrates the results presented in Chapters 4 and 5. The many different results presented in those chapters are reviewed and consolidated to provide a reasoned basis for evaluating the performance of the disposal facility. This interpretation provides a rational basis to conclude that the performance of the disposal facility has been completely addressed, the analysis is logically interpreted, the results are correct representations of the facility performance, and the results are sufficiently rigorous. This chapter also provides information for establishing waste acceptance criteria.

6.2 INTEGRATION OF RESULTS

Section 7.1 compares the results to the performance objectives in detail. However, based on the results from Chapters 4 and 5, a few general principles stand out:

- All performance objectives associated with release and migration of radionuclide to the point of compliance are met with a wide margin (ratio of predicted value to performance objectives). The intruder dose performance objective is met with a smaller margin (which is expected to increase based on recent knowledge of the separation of Sn-126 in the Waste Treatment Plant (WTP). The performance goals associated with release and migration of hazardous chemicals to the point of compliance is met with an even wider margin than met by radionuclides.
- A few key radionuclides are important (^{99}Tc, ^{129}I, ^{237}Np, and uranium isotopes for the groundwater pathway and ^{126}Sn, ^{239}Pu, and ^{241}Am for the inadvertent intruder scenario)
- The waste form release rate determines the temporal shape and is a major driver in determining the magnitude of the groundwater and human health impacts
- Infiltration rates are important because they affect waste form release rates and travel times
- The disposal system design can affect infiltration rates into the disposal facility
- The placement of the disposal facility over the ancient Columbia River bed increases the amount of dilution relative to other locations
- The performance objectives are by and large satisfied even if extreme assumptions are used.

As in all previous Hanford Site assessments, the mobile, long-lived radionuclides are most important for the groundwater pathway. Although the relative importance depends on the performance measure being used (beta-photon dose from drinking water, alpha concentration in drinking water, all-pathways), it is clear that ^{99}Tc, ^{129}I, uranium isotopes, and ^{237}Np are the most

important isotopes. Table 6-1 provides the information for the all-pathways dose. Section 4.3.6 provides more information.

Table 6-1. Important Radioisotopes in Groundwater Pathway.

	Groundwater All-pathways Dose (mrem/year)		
	1,000* Years	10,000 years	100,000 years
^{99}Tc	0.0000552 (71%)	0.0262 (37%)	0.0273 (4%)
^{129}I	0.0000224 (29%)	0.0110 (16%)	0.0153 (3%)
^{231}Pa	0.0	0.00048 (1%)	0.0219 (4%)
^{233}U	0.0	0.0031 (4%)	0.0456 (8%)
^{234}U	0.0	0.00104 (1%)	0.0177 (3%)
^{238}U	0.0	0.00108 (2%)	0.0180 (3%)
^{237}Np	0.0	0.0270 (39%)	0.4388 (75%)
Other	0.0000002	0.0001 (0%)	0.0014 (0%)
Total	0.0000778	0.0700	0.588

*Time of compliance (percentage contribution is given in parenthesis)

For the inadvertent intruder scenario, a different set of radioisotopes is important. As shown in Table 6-2, the isotopes that are important vary with time (also see Figure 5-2). At short times, the important radioisotopes are ^{90}Sr and ^{137}Cs, which have half lives around 30 years. However, by the time of compliance (500 years), these two isotopes have decayed and the long-lived isotopes are important (^{126}Sn, ^{239}Pu, and ^{241}Am) with ^{126}Sn contributing over two-thirds of the total dose. A fuller discussion is given in Section 5.4.

Table 6-2. Important Radioisotopes in Homesteader Inadvertent Intruder Scenario.

	Dose to Inadvertent Homesteader (mrem/year)		
	100 Years	500 years *	1,000 years
^{90}Sr	68.4 (13%)	0.00 (0%)	0.00
^{126}Sn	7.0 (1%)	7.00 (70%)	6.99 (76%)
^{137}Cs	461. (85%)	0.04 (0%)	0.00
^{237}Np	0.4	0.35 (3%)	0.35 (4%)
^{239}Pu	0.6	0.61 (6%)	0.60 (7%)
^{240}Pu	0.1	0.10 (1%)	0.10 (1%)
^{241}Am	3.3 (1%)	1.76 (17%)	0.80 (9%)
Others	0.6	0.31 (3%)	0.30 (3%)
Total	541.4	10.17	9.14

*Time of compliance (percentage contribution is given in parenthesis)

As found in the 1998 ILAW performance assessment (Mann 1998a), the time structure of the waste form release drives the time dependence of the main impacts from the groundwater pathway. This is not difficult to understand because the travel times in the vadose zone and in the groundwater are so much shorter than the release times associated with the waste form. Basically, the disposal facility provides a steady source to the vadose zone (and subsequently to the groundwater) with these latter two zones just diluting and delaying the arrival of the contaminants at the point of observation in the groundwater. Thus, waste form release continues to be a major driver in the groundwater pathway. For more information, see Section 4.3.3 on the waste form release results for the base analysis case and Section 4.3.6 on the integration of results for the base analysis case.

Infiltration rates are important as they affect waste form release rates and travel times. Figure 4-4 shows a four order of magnitude increase in the waste form release rate as the infiltration rates increases from 0.1 to 10 mm/year. Figure 4-35 shows that the travel time through the vadose zone to the groundwater decreases as the infiltration increases. At 10,000 years, the difference in contaminant flux to the groundwater is about 5 orders of magnitude for infiltration rates varying between 0.1 and 50 mm/year. The difference at shorter times is even larger. For example at 1,000 years (the time of compliance), the contaminant flux from 50 mm/year infiltration is about 300 times the flux corresponding to infiltration of 4.2 mm/year and well over 5 orders of magnitude larger than the contaminant flux corresponding to 0.9 mm/year.

Disposal design can affect infiltration rates into the disposal facility. This is most easily seen by comparing the results from the base analysis case, which assumes a natural recharge of 4.2 mm/year, and the best estimate case, which assumes a sand-gravel capillary barrier. Such a barrier greatly reduces infiltration and hence release rate. When comparing the normalized flux to the groundwater for the base analysis and best estimate cases in Figure 4-21, the presence of the sand gravel capillary barrier could reduce the contaminant flux to groundwater by over four orders of magnitude.

The relationship of disposal facility placement to the ancient Columbia River bed affects the amount of dilution. The 1998 ILAW performance assessment ignored the presence of the ancient Columbia River Bed. The higher flow through this region and the difference in the facility footprints between the 1998 ILAW PA and this analysis has led to well intercept factors that are approximately a factor of 10 lower for this assessment. Also this geological structure tends to minimize the spreading of the contaminant plume laterally. Therefore, the WIFs at 100 m and 1,000 m are similar (see Table 4-2).

Estimated impacts from two extreme cases demonstrated that even using extreme assumptions do not lead to levels greatly higher than performance objectives. The first case assumed that the waste form released all contaminants over the period of 1 year. This is an increase in the release rate of the waste form by a factor of approximately 1,000,000. The resulting peak estimated impacts (see Table 4-19) were approximately a factor of 1,000 higher than the estimated impacts at 10,000 years for the base analysis case. However, the peak estimated impacts are only a factor of four above the groundwater performance objectives. The second case assumed the disposal facility was located just above the unconfined aquifer (i.e., there was no vadose zone). The waste form was assumed to release its contaminants as

estimated for the base analysis case. The resulting estimated impacts for the groundwater scenario were approximately a factor of 100 higher than the estimated impacts at 10,000 years for the base analysis case. The estimated impacts remained below the groundwater performance objectives. The major radionuclide contributors for this case assuming no vadose zone are anticipated to be chemically retarded in their transport through the vadose zone. Specifically ^{90}Sr and ^{137}Cs are the major contributors to the beta-photon drinking water dose, with the peak occurring at 70 years after facility closure, and have measured $K_d = 10$ and 80 mL/g, respectively. Similarly, the major contributor to the all-pathway dose at 10,000 years after facility closure is ^{239}Pu which has a measured $K_d = 200$ mL/g.

6.3 VERIFICATION OF PERFORMANCE ASSESSMENT RESULTS

This section discusses the basis to conclude that the performance of the ILAW disposal action has been completely addressed, the analysis is logically interpreted, the results are representative of the facility performance, and the results are sufficiently rigorous.

The completeness of the ILAW performance assessment activity can be judged on the effort to determine not only the best estimate case but also other cases that might affect the impacts of waste disposal. An important comment (See Section 1.6.4) is that the public not only wants to know what is expected to happen, but also how bad the consequences could be if these expectations are wrong. Thus, in the creation of the data packages and simulations for this performance assessment (Mann/Puigh 2000a and Puigh 2001) investigations were included to determine basic processes and whether the parameters being measured and the associated models are appropriate. The best example of this is the strategy for glass analysis (See Appendix E). In this document, numerous sensitivity studies (see Section 3.5.5) were conducted to determine how possible alternative choices could affect the results. Other examples are placement of the disposal facility (Section 4.7.2), design of the disposal facility (Sections 4.37 through 4.3.11, Section 4.5.5, 4.5.6), alternative glass formulation (Section 4.5.7), and different inventories (Section 4.8.3). Other cases were run to determine the dependence of the results based on model choices. Examples are waste release models (Section 4.5.2 and 4.5.3) and the presence of clastic dikes in the vadose zone (Section 4.6.3). Still other cases were run to determine the sensitivity of the results to uncertainties in parameters. Examples are infiltration rates (Sections 4.5.4 and 4.6.2), vadose zone parameters (Section 4.6.3), and groundwater flow (Sections 4.7.4 and 4.7.5).

That this analysis is logically interpreted can be judged by reviewing the simple overall approach presented in Section 3.3. Calculations involving the air pathway and the inadvertent intruder scenarios are straightforward. In concept, the groundwater pathway is straightforward (water enters the disposal facility, waste degrades because of water contact, the water carries the released contaminants to groundwater which carries the contaminants to the point of compliance). Despite the straightforwardness of the concept, the models used in each step are fairly complicated. However, for each step, simple models (no barrier case, forward rate for glass degradation, mobile contaminant transport, and simple groundwater dilution) are provided to show that the complicated models provide appropriate results.

These results are not the exact representations of the facility performance because the disposal facility has not been designed and the waste form has not been finally specified.

However, through the use of conservative as well as best estimate cases, this performance shows how likely candidate designs and waste forms will perform. In addition, sensitivity cases were run to determine the effect of various choices. The models and codes used are well established (Section 3.5) and the data establishing the values for the parameters in the models and codes are based on extensive sets of measurements and have been thoroughly documented (Mann/Puigh 2000a). For important features like glass degradation and groundwater flow, calculations have been compared to laboratory and field measurements.

This PA uses state-of-the art data, methods, and codes, which cause the results to be sufficiently rigorous. Moreover, as stated earlier, by comparing these results to those for simpler cases and to bounding cases, any limitations of the methods or codes would have an insignificant consequence to the overall interpretation.

6.4 BASIS FOR WASTE ACCEPTANCE LIMITS

Because waste acceptance limits are determined based on a comparison to performance objectives, the discussion of actual waste acceptance limits is postponed to Section 7.6.1. These limits will be determined by comparing the results of the base analysis case to the performance objectives for each radioisotope or chemical material.

7.0 PERFORMANCE EVALUATION

7.1 OVERVIEW

This chapter compares the estimated impacts found in Chapters 4 and 5 with the performance objectives established in Chapter 1 (Section 7.2) as well as the sensitivity of the results to key parameters (Section 7.3). Section 7.4 discusses the conservatisms and caveats associated with the analysis. This chapter also defines those requirements (Section 7.6), both on the waste form and on the disposal facility design, that are derived from long-term performance and shows that meeting these requirements does not require heroic effort and so the requirements are likely to be achieved. Section 7.7 describes the reasons why the results from this assessment differ from the assessment document in the 1998 ILAW PA (Mann 1998a). The chapter concludes by describing the work planned to update the information contained in this PA.

7.2 COMPARISON OF ESTIMATED IMPACTS TO PERFORMANCE

This section compares the estimated impacts to the performance objectives for each area of protection cited in Section 1.3:

- Protection of the general public
- Protection of the inadvertent intruder
- Protection of groundwater resources
- Protection of surface water resources
- Protection of air resources.

The inadvertent intruder estimated impacts depend on inventory and facility design, and can be mitigated to some extent operationally. The estimated impacts for the other performance objectives (except for air resources) depend on inventory, waste form release rate, facility design, and groundwater flow rate.

7.2.1 Protection of General Public

Table 7-1 compares the performance objectives for protecting the general public with the results from the base analysis and best estimate cases. The estimated all-pathways doses are significantly lower than the performance objectives during the first 10,000 years. The DOE time of compliance is 1,000 years.

Table 7-1. Comparison of Estimated Impacts with Performance Objectives for Protecting the Public. The DOE time of compliance is 1,000 years.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years	Estimated Impact at 10,000 years	
			1998 ILAW PA	Present Results
All-pathways [mrem in a year]	25.0			
Base analysis case		0.000078	6.4	0.070
Best estimate case		1.7×10^{-10}	nc*	1.3×10^{-6}

* nc = "not calculated" in the 1998 ILAW PA

7.2.2 Protection of Inadvertent Intruders

Table 7-2 compares the estimated impacts to the performance objectives for protecting the inadvertent intruder. The time of compliance starts at 500 years after closure. The acute exposure performance objective is met by a factor greater than 500. ^{126}Sn is the most important radionuclide. The continuous exposure performance objective is met by a factor of approximately four for the base analysis case. ^{126}Sn, ^{241}Am, and ^{239}Pu are the major contributors. These results are similar in magnitude to those found in the ILAW PA (Mann 1998a).

Table 7-2. Comparison of Estimated Impacts with Performance Objectives for Protecting the Inadvertent Intruder. The time of compliance starts at 500 years.

Performance Measure	Performance Objective	Estimated Impact at 500 years	
		1998 ILAW PA	Present Results
Acute exposure [mrem]	500.0	5.5	0.76
Continuous exposure [mrem in a year]	100.0	27.5	10.2

The estimated impacts for the inadvertent intruder can be mitigated through operational controls based on projected container inventories. Such operational controls will be better defined as the project matures.

7.2.3 Protection of Groundwater Resources

Table 7-3 compares the estimated impacts to the performance objectives for protecting the groundwater resources. At the DOE time of compliance (1,000 years) and the point of compliance (at a well 100 m downgradient of the disposal facility), the groundwater impacts are not significant. At 10,000 years the estimated impact is approximately a factor of 400 less than the performance objectives for beta-photon emitters and the alpha-emitting radionuclides for the base analysis case.

The concentration of radium is insignificant. The most important drivers are the inventories of technetium, iodine, neptunium, and uranium, the release rate from the waste form, and the amount of mixing in the aquifer. Retardation of uranium isotopes as they migrate through the natural vadose zone is important in achieving the alpha-emitting radionuclides performance objective. The anticipated retardation of the uranium isotopes through any concrete associated with the engineered facility has not been included in these estimates. (See Section 4.6.5 for a description of this case and its results.)

Table 7-3. Comparison of Estimated Impacts with Performance Objectives for Protecting Groundwater Resources. The DOE time of compliance is 1,000 years. The point of compliance is a well 100 m downgradient of the facility.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years	Estimated Impact at 10,000 years	
			1998 ILAW PA	Present Results
βγ Emitters [mrem/year]	4.0			
Base analysis case		2.1×10^{-5}	2.0	0.0102
Best estimate case		4.7×10^{-11}	nc*	3.5×10^{-7}
Alpha-emitters [pCi/L]	15.0			
Base analysis case		1.0×10^{-16}	1.7	0.034
Best estimate case		2.3×10^{-22}	nc*	4.8×10^{-8}
Ra [pCi/L]	5.0			
Either case		0.0	<0.001	<0.001

* nc = "not calculated" in the 1998 ILAW PA

7.2.4 Protection of Surface Water Resources

Table 7-4 compares the estimated impacts to the performance objectives for protecting the surface water resources. The DOE time of compliance is 1,000 years and the point of compliance is at a well intercepting the groundwater just before it mixes with the Columbia River. The estimated impacts are approximately three orders of magnitude lower than the performance objectives for the base analysis case. The estimated impacts at a well just before the river are conservative with respect to the quality of the river water.

Table 7-4. Comparison of Estimated Impacts with Performance Objectives for Protecting Surface Water Resources (Base Analysis Case). The DOE time of compliance is 1,000 years. The point of compliance is a well intercepting the groundwater before entering the Columbia River.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years	Estimated Impact at 10,000 years	
			1998 ILAW PA	Present Results
All-Pathways [mrem/y]	25.0	7.3×10^{-6}	0.22	0.0065
βγ Emitters [mrem/y]	1.0	2.0×10^{-6}	0.070	0.00095
Alpha Emitters [pCi/L]	15.0	1.0×10^{-17}	0.058	0.0032
Ra [pCi/L]	3.0	0.0	<0.001	<0.001

7.2.5 Protection of Air Resources

Table 7-5 compares the estimated impacts to the performance objectives for protecting air resources. The DOE time of compliance is 1,000 years and the point of compliance is just above the disposal facility. The estimated impacts are over three orders of magnitude lower than the performance objectives.

Table 7-5. Comparison of Estimated Impacts with Performance Objectives for Protecting Air Resources. The DOE time of compliance is 1,000 years. The point of compliance is just above the disposal facility.

Performance Measure	Performance Objective	Estimated Impact at 1,000 years	
		1998 ILAW PA	Present Results
Radon [pCi m^{-2} $second^{-1}$]	20.0	<0.001	<0.96
Other radionuclides (^{3}H and ^{14}C) [mrem in a y]	10.0	$<10^{-8}$	$<6 \times 10^{-3}$

7.2.6 Summary

All of the estimated impacts easily meet the performance objectives set out in Section 1.3 for the remote-handled waste trench base analysis case. The estimated all-pathways dose, beta-photon drinking water dose, and concentration of alpha-emitting radionuclides in groundwater for the base analysis case are more than a factor of 300 lower than the corresponding performance objective at 10,000 years after facility closure (2030). This margin increases by well over an order of magnitude if the time of compliance of 1,000 years is used. Using the results of the best estimate case (where a subsurface sand-gravel capillary barrier is in place), the margin increases by orders of magnitude more. The

inadvertent intruder continuous exposure doses are estimated to be a factor of approximately four below the performance objective.

7.3 PERFORMANCE SENSITIVITY TO KEY PARAMETER UNCERTAINTIES

The key uncertainties of this analysis are as follows:

- Disposal facility design and layout
- Waste form performance
- Groundwater flow
- Inventory.

7.3.1 Disposal Facility Design And Layout

Disposal facility design and layout affect impacts in two ways. They modify the amount of moisture available to degrade the waste form and that is available to carry the released contaminants to the groundwater. Facility layout also affects the amount of groundwater available for diluting the contaminants reaching groundwater.

Infiltration into the disposal facility has a major effect on the results of the analysis. Most human-made features are expected to degrade relatively quickly compared to the waste form. Nonetheless, through the use of natural materials, certain features, such as the subsurface sand-gravel capillary barrier or vertical barriers, might be expected to last a long time. In particular, the capillary barrier has a major impact on the amount of moisture entering the actual disposal facility and hence on the effect of that moisture. By comparing the results of the base analysis case and the best estimate case, the sand-gravel capillary barrier is estimated to reduce the impacts by many orders of magnitude. (See Sections 4.3.7, 4.5.4, and 4.6.2.)

The presence of the ancestral Columbia River channel has a significant effect of focusing the groundwater flow beneath the disposal facility. Because relatively few boreholes define this channel, its exact shape is uncertain. However, even if the channel is replaced by the Ringold Formation (the relatively tight geologic unit through which the ancestral Columbia River cuts), the impacts increase by only about an order of magnitude at a well 100 m down gradient and by even less farther out. The effects of small changes in the channel shape are minor, as shown in the relatively small change (50 percent) in impacts if the disposal facility is moved to the south end of the disposal site. Section 4.7 has a fuller description.

7.3.2 Waste Form Performance

The degradation of the waste form sets the time structure of system performance and, so has a major effect on the estimated impacts of the disposal action. A wide variety of sensitivity and uncertainty cases were run. Even if the glass degrades at the fastest rate possible (the "forward rate"), the impacts increase by only an order of magnitude from the

base analysis case. Increasing the moisture that the glass contacts does have an impact (a factor 8 increase in the glass dissolution rate for a factor of 2.5 increase in infiltration rate at 1,000 years, but only a factor of 2.6 at 10,000 years). However, the base analysis case uses an infiltration rate (4.2 mm/year) corresponding to the wetter of the two natural conditions found on the disposal site.

These analyses are based on the very extensive database of the LAWABP1 glass. If a different glass is used, the results may change. Some glasses have been tested that would perform much more poorly than LAWABP1; however, many glass compositions are available that perform better than LAWABP1. (See Section 7.5, Figure 7-2, for a fuller discussion of the uncertainty in glass performance.)

7.3.3 Groundwater Flow

Because the amount of water entering the aquifer with the contaminants is so small compared to the groundwater flow, the estimated impacts from this disposal action are inversely proportional to the groundwater flow.

The groundwater flow beneath the Hanford Site is presently changing. During the Site's early history, large amounts of water (including radioactive and hazardous waste) were discharged directly to the ground. This caused the groundwater table to form mounds that are now gradually dissipating. Thus, present groundwater flow measurements are not indicative of future groundwater flow.

The estimates of groundwater flow used here are based on the Hanford Site groundwater model. This model was used in the Hanford Site Composite Analysis (Kincaid 1998). The sensitivity of the estimated impacts to this model was investigated by a large number of sensitivity studies (Section 4.7). In general, sensitivity is relatively low except for the case where the ancestral Columbia River channel is replaced by the Ringold Formation. In this case the impacts locally increase by an order of magnitude. However, small changes in the exact shape of this channel should not greatly affect the estimated impacts.

7.3.4 Inventory

Only a few contaminants (^{99}Tc, ^{126}Sn, ^{129}I, ^{137}Cs, uranium isotopes, ^{237}Np, ^{239}Pu, and ^{241}Am) drive the results. The inventory in the ILAW packages results from a separation process on waste from the Hanford Site tanks. For the purposes of this performance assessment, Hanford Site tank waste is reasonably well characterized. The separation step, separating the tank waste into low-activity waste and high-level waste, still is being designed.

^{99}Tc drives the groundwater results, while ^{137}Cs drives the inadvertent intruder dose at times of less than 200 years. Both of these isotopes have separation steps in the tank waste treatment process. This assessment uses the requirements of the tank treatment contract (DOE/BNFL 1998) and the separations information provided in Wootan (1999). However, other cases are examined and the changes are relatively small.

The amount of ^{126}Sn and ^{129}I is relatively uncertain. Whether tin will be in the low-activity waste stream or the high-level waste stream is unclear. To be conservative, tin was assumed to be in the low-activity waste stream for this analysis. The amount of iodine in the waste tanks also is poorly known. Most of the iodine is believed to have escaped through the air pathway during the initial chemical treatment of the fuel many decades ago. To be conservative, all the iodine was assumed to go to the low-activity stream. However, because of its high volatility, 90 percent of the iodine does not go into the glass during the treatment process but instead will be trapped on air filters that will be disposed in the onsite solid waste disposal sites.

The amount and chemistry of other key materials (uranium isotopes, ^{237}Np, ^{239}Pu, and ^{241}Am) are quite well known because these materials were the main products of the Hanford Site's original mission. Small changes do not have significant effects.

7.3.5 Other Factors

A variety of other factors could affect the estimated results. However, such factors are investigated in the sensitivity cases. The biggest changes not yet considered in this section are retardation factors. The present values are based on extensive site-specific experiments. Significant changes are not expected from this data set.

7.4 CONSERVATISMS AND CAVEATS

7.4.1 Conservatisms

The intent of this assessment was to be as realistic and rigorous as possible. However, the presence of some still-unmade decisions (disposal facility design, waste form composition, pretreatment separations) has created the need for some conservatism.

The present disposal facility conceptual design has a subsurface sand-capillary barrier. The exact specifications for this disposal facility feature have not been determined. Thus, the base analysis case assumes no capillary barrier, resulting in a significant conservatism. Moreover, the base analysis case assumes a natural recharge equivalent to the wetter of the two soil types, thus causing more moisture flux into the disposal facility. Because estimated performance depends greatly on this infiltration, actual results should be less. However, the actual performance will depend on the performance of the degraded surface barrier, which could be composed entirely of Burbank sandy loam, the wetter of the two soil types.

All but one of the waste form calculations were one dimensional. This has the effect of driving all moisture through the disposal facility's glass-containing regions. In actuality, water will be diverted around the glass waste form because the filler material has much higher hydraulic conductivity than the waste form. Also, all glass was assumed to be LAWABP1 with 20-weight percent waste loading. However, based on current plans, little if any of the glass is likely to have such a high contaminant loading and the average loading is likely to be less. As will be seen in Section 7.5, which discusses glass performance, lower

waste loading means better waste form performance because of lower pH conditions in the trench.

The separation step in the treatment process has not yet been determined. To be conservative, the maximum likely amount of ILAW inventory was assumed to be in the ILAW packages.

7.4.2 Caveats

This effort, as is true for the rest of the ILAW PA effort, is being performed before all decisions concerning ILAW have been made. The treatment plant designer still must decide on the waste form composition that will be used. Although the flow sheet is becoming finalized, its details still must be transmitted to the ILAW PA activity. Similarly, the detail design for the disposal facility does not yet exist. Finally, although the amount of disposal site-specific information has increased, more data are still needed.

This analysis is based on a waste form composition in the composition space that BNFL Inc. (the original treatment plant designer) has chosen. However, the treatment plant designer will not select its glass waste form composition for some months. Similarly, the treatment flow sheets used here are those developed by the Hanford Site contractor. Although the effects of these flow sheet decisions are expected to be small, they still must be investigated to determine the size of the change in the environmental impacts. In particular, once the waste form composition and process range are known, significant waste form testing (similar to that performed for LAWABP1) will be conducted.

The present design for the disposal facility is based on conceptual designs based on the existing mixed-waste trench at the Hanford Site. As detailed design occurs, the dimensions are expected to change and the materials will be more closely specified. Again, the impacts of these changes are expected to be small.

Finally, more geotechnical data (both from the ILAW disposal site and for the Hanford Site as part of the Hanford Site Groundwater/Vadose Zone Integration Project) will be obtained. These data will be incorporated to better define conceptual models and the parameters used to implement those models. Based on the 1998 ILAW PA, the effect of the new data will be a better understanding of the flow and transport, but relatively little change in values is expected.

The impacts from the decisions discussed above will be addressed in the performance assessment annual summaries and performance assessment revisions. Both of these sets of documents are required by the *Maintenance Plan for the Hanford Immobilized Low-Activity Tank Waste Performance Assessment* (DOE/ORP 2000a) as well as the DOE order on radioactive waste management (DOE 1999b).

7.5 UNCERTAINTIES REGARDING GLASS PERFORMANCE

The calculations and long-term performance results discussed in earlier sections of this document are based on a detailed analysis of the release behavior of LAWABP1 glass. However, it is a virtual certainty that the produced ILAW glasses will not have this specific composition. Consequently, it is important to assess the likelihood that the glass waste forms that will be produced will have long-term durability characteristics approximating those of LAWABP1 glass. Unfortunately, doing so is impossible quantitatively for the following reasons. Specific glass formulations for ILAW disposal have not been selected for production. Insufficient experimental data are available to perform STORM simulations with the glasses, even if the compositions themselves were available.

As an intermediate step, the relative performance of BNFL Inc.-type glass compositions can be compared in highly accelerated laboratory tests designed to elucidate the long-term behavior of the materials on a practicable time scale (McGrail, et al. 1998b). Two experimental methods are principally used for this purpose (McGrail et al. 1998b), the vapor hydration test (VHT) and the pressurized unsaturated flow (PUF) test. Briefly, in the VHT, monolithic samples are exposed to saturated water vapor at temperatures typically of 100°C to 300°C in a sealed vessel. This environment greatly accelerates the progression of glass corrosion by water and can result in the formation of alteration phases. The principal uses of the test are as follows:

- As a screening tool to quickly determine if a glass is likely to corrode at an extreme rate
- As a convenient means of generating alteration phases for analysis within a short period
- For a measure of the alteration rate at elevated temperatures.

In contrast, the patented[1] PUF test is an open-system test where water flows through a bed of coarsely ground glass under conditions of partial hydraulic saturation (McGrail et al. 1997). A computer control system stores test data to disk from several thermocouples, pressure sensors, inline sensors for effluent pH and conductivity, and column weight from an electronic strain gauge to accurately track water mass balance and saturation level. Experience in running PUF tests with a number of different ILAW glass compositions has proven the method to be highly effective in accelerating the progression of the glass corrosion process into the so-called "Stage 3" regime representative of longer-term corrosion behavior, and detecting glasses that are unstable with respect to secondary phases that form as a result of the glass-water reaction processes during the test.

A matrix of 56 glass compositions was subjected to VHTs at 200°C for sufficiently long periods to obtain a statistically meaningful measure of the glass corrosion rate (Vienna 2000). The glasses varied the concentrations of SiO_2, Al_2O_3, B_2O_3, Fe_2O_3, TiO_2, ZnO, ZrO_2,

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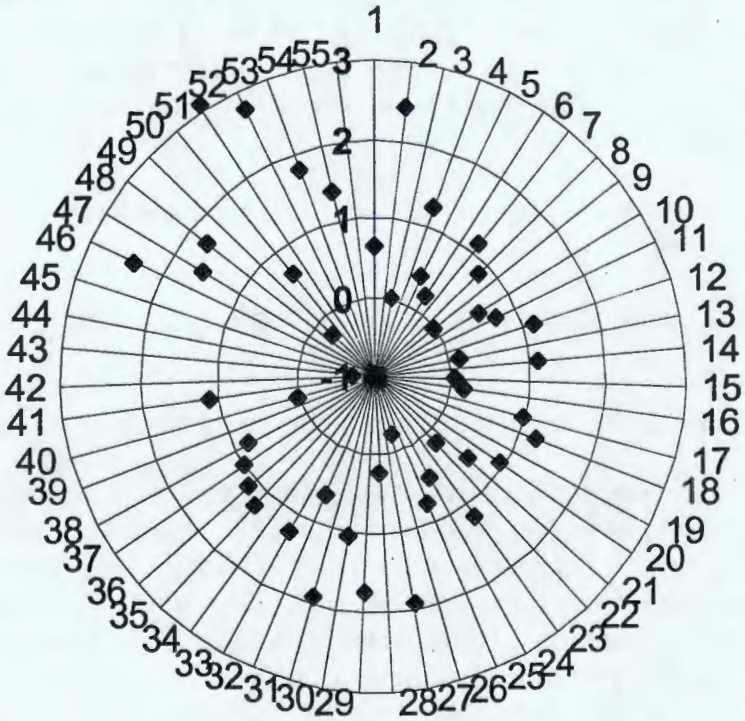
[1] Patent #5974859, "Method and Apparatus for Measuring Coupled Flow and Reaction Processes," 1999.

MgO, and Na_2O across a wide composition range that covers, with high probability, the expected processing composition range of the ILAW producer. The test matrix was designed in collaboration with staff at the Catholic University of America who are principally responsible for ILAW waste form development to ensure that the selected components and ranges were relevant to glasses under current development. For details on the specific glass compositions involved, please see Vienna (2000). In Figure 7-1, we plot the logarithm of the measured VHT corrosion rate for 50 glass compositions. (Note that results for six of the test glasses were not yet available.) Immediately obvious from the plot is that a large fraction of the test glasses have corrosion rates less than 10 g/(m² d). This result was quite unexpected because the aggressive, high-temperature conditions of the VHT were anticipated to produce high corrosion rates for a significantly larger fraction of the test glasses.

Figure 7-1. Radial Distribution Plot of 200°C VHT Corrosion Rates for HLP Series of ILAW Glasses. Radial coordinates are $\log_{10}$ corrosion rate, g/(m² d).

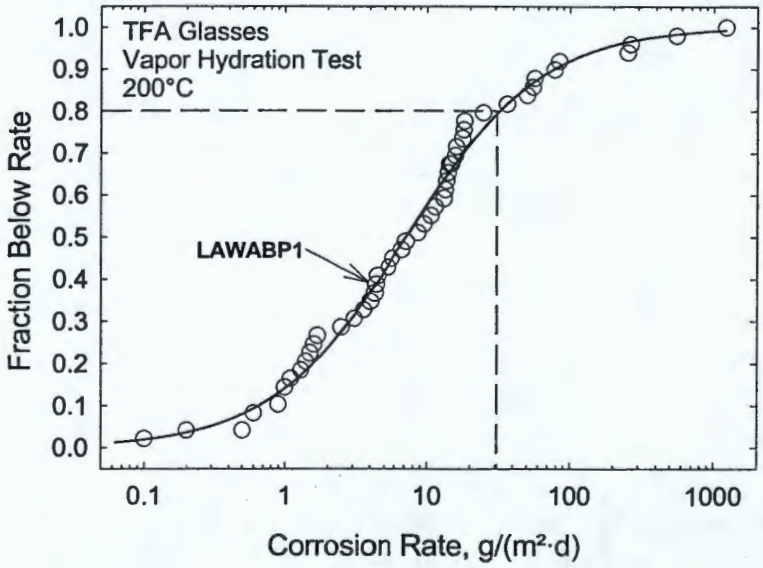


To more quantitatively analyze the results, the VHT corrosion rate data have been replotted in the form of a cumulative distribution function, as shown in Figure 7-2 for glasses studied under the Tank Focus Area (TFA). The measured 200°C VHT corrosion rate for LAWABP1 glass is 4.4 g/(m² d) and the corresponding data point is highlighted in Figure 7-2. This glass is near the midpoint of the distribution (half of the data set have higher rate and half lower) of 7.2 g/(m² d). A full 80 percent of the tested glasses have 200°C VHT

corrosion rates less than 30 g/(m² d). This is about eight times faster than the VHT rate for LAWABP1 glass. However, a glass reacting eight times faster than LAWABP1 still would fall well within the margin of safety available to meet groundwater pathway performance objectives, based on the data in Table 7-9.

Figure 7-2. Cumulative Distribution Plot of 200°C VHT Corrosion Rates for HLP Series of ILAW Glasses.

The data were fit to a three-parameter logistic function of the form $y = a/[1+(x/x_o)^b]$.



Because of the much greater complexity of the hardware and support equipment, fewer experimental data exist in terms of compositions tested via the PUF system. The latest available data relevant to BNFL Inc. compositions are shown in Figure 7-3. The lines in the figure were computed using a 4-point moving average for the HAN28, LAWA23, and LAWA33 glasses and a 10-point moving average for the HLP-10, HLP-31, and LAWABP1 glasses. A comparison of the peak dissolution rate observed in PUF tests versus the dissolution rate estimated in VHTs at 200°C is plotted in Figure 7-4. The peak dissolution rate was used from the PUF tests because, for the glasses with high dissolution rates, the apparent corrosion decreases with time as the total unreacted glass surface area decreases. This is not taken into account in the corrosion rate calculation.

The results suggest a good correlation between the VHT and PUF test results ($R^2 = 0.91$). Similar secondary phases formed in both types of tests, which is probably why a correlation exists between the results. The available data obviously still are very limited. However, the VHT appears to be a good indicator of glass performance in the PUF test, and both accelerated tests are providing a consistent picture of long-term performance of ILAW glasses as a function of glass composition. These results should be confirmed as additional PUF test data are developed on more ILAW glasses. Meanwhile, it appears virtually certain that glasses can be formulated and manufactured that will meet performance objectives for disposal of low-activity tank waste.

Figure 7-3. Comparison of Glass Corrosion Rate in PUF Tests at 99°C and 2 mL/d.

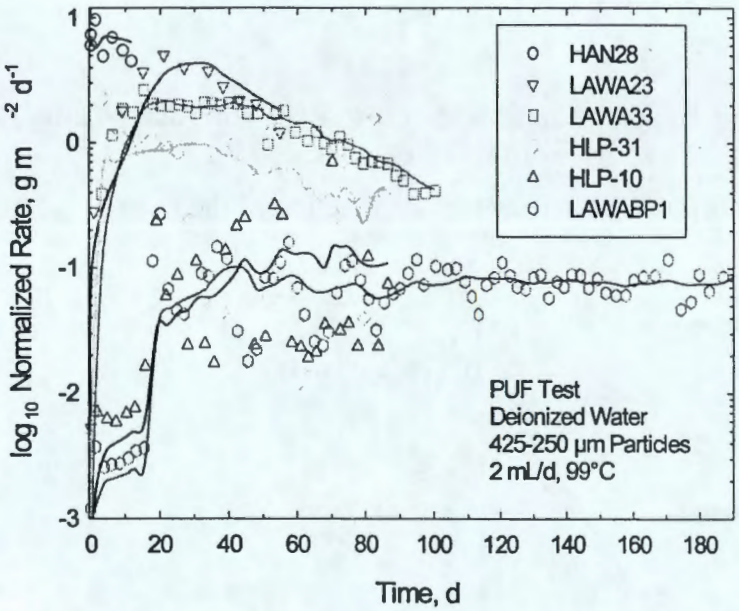
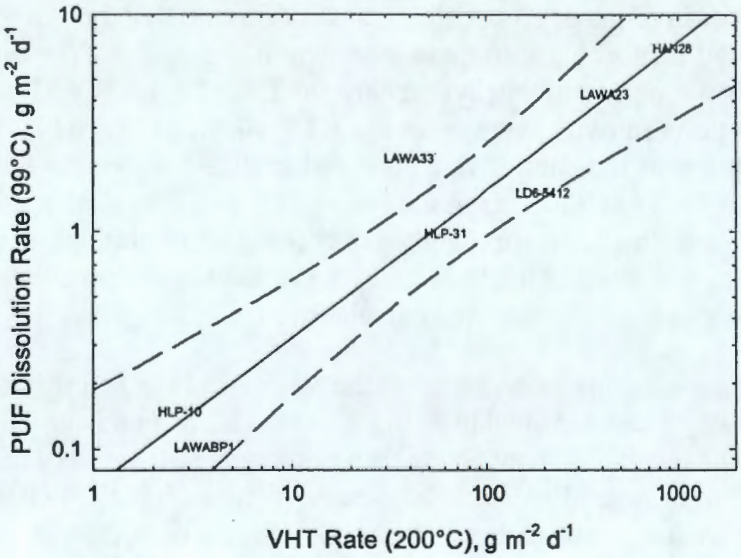


Figure 7-4. Comparison of Glass Corrosion Rate in PUF Tests at 99°C and VHT Tests at 200°C.

Solid line is the regressed fit and the dashed lines are the 95 percent confidence interval.



7.6 REQUIREMENTS SET BY PERFORMANCE ASSESSMENT

7.6.1 Introduction

The analysis presented so far was designed to establish an understanding of how the engineered and natural systems behave. Based on this understanding, this section derives the waste form and disposal facility requirements based on long-term human health and safety and environmental performance objectives. Section 7.6.2 and 7.6.3 derive the equations and restrictions based on protecting the inadvertent intruder and groundwater. Sections 7.6.4 and 7.6.5 describe how the derived restrictions apply to the waste form and disposal facilities. Section 7.6.6 describes why the requirements set in the preceding sections are likely to be met.

Data and methods used to establish requirements on the waste form and disposal facilities come mainly from the data and methods used in the base analysis case (Chapter 3) and the base intruder cases (Section 5.3). However, the performance goals for groundwater protection are selected to be more conservative than the performance objectives described in Section 1.6. This will provide an additional margin of safety early in the design process so that engineering tradeoffs, if needed, can more easily be made in a complex, non-linear system.

Only the homesteader inadvertent intruder and groundwater scenarios were used to establish requirements for the waste form and the disposal facilities. Doses from the driller inadvertent intruder, the surface water, and the air release scenarios when compared to their performance objectives are much less restrictive than those from the groundwater and homesteader scenarios. The performance goals are given in Table 7-6.

Table 7-6. Performance Goals for Requirement Cases.

Performance Measure	Requirement Point	Performance Objective	Performance Goal
Continuous inadvertent intruder dose	Disposal Facility	100 mrem/year @ 500 years	100 mrem/year @ 500 years
All-pathways dose	Well 100 m downgradient	25 mrem/year @ 1,000 years	5 mrem/year @ 10,000 years
Beta-gamma drinking water dose	Well 100 m downgradient	4 mrem/year @ 1,000 years	1 mrem/year @ 10,000 years
Alpha-emitter concentration	Well 100 m downgradient	15 pCi/L @ 1,000 years	5 pCi/L @ 10,000 years

From these performance goals, requirements can be derived for parameters more under the control of designers (waste treatment plant specifications and disposal facility specifications) and of operators (waste treatment plant and disposal facility). Such types of requirements are shown in Table 7-7.

Table 7-7. Types of Operation Parameters to be Restricted by PA.

Type of Requirement	Source
Limits on product of inventory and contaminant release rate	Present analysis: Table 7-9
Limits on inventory concentration times effective height	Present analysis: Table 7-8
Limits on inventory concentration	Class C limits [10 CFR 61] Contract (DOE/ORP 2001)
Limits on smearable inventory	Present analysis: Table 7-10 Contract (DOE/ORP 2001)

7.6.2 Requirement Development Based on Inadvertent Intruder Protection.

For the protection of the homesteader, the dose to the inadvertent intruder depends on the amount of each contaminant retrieved (A_i) multiplied by the effective dosimetry parameter (d_i^h), summed over all the contaminants i. Thus, the intruder dose (D^h) must be less than the performance goal established in Table 7-6 (D^h_{max}),

$$D^h = \sum_i k_1 A_i d_i^h \le D^h_{max} \tag{7.1}$$

where

D^h = the dose to the inadvertent intruder (mrem/year)

k_1 = a constant that includes the effect of diluting the waste in the garden soil ($1/m^3$)

A_i = amount of contaminant i retrieved (Ci)

d_i^h = the dosimetry factor relating response to concentration of contaminant i in the inadvertent homesteader scenario [(mrem/year)/(Ci/m^3)], and

D^h_{max} = the maximum dose allowable in the homesteader scenario (mrem/year).

The amount of each contaminant retrieved depends on the volume of the waste retrieved times the concentration of the waste. Noting that the waste is homogeneous in each of the ILAW packages, the amount of each contaminant retrieved is

$$A_i = k_2 \sum_j \left[\frac{I_{i,j}}{V_j} \right] H_j \tag{7.2}$$

where the sum of j is the sum over packages in a vertical column,

k_2 = area of the hole going through the waste (m^2)

$I_{i,j}$ = the inventory of contaminant i in container j (Ci)

V_j = the volume of waste in container j (m^3), and

H_j = the height of waste within container j (m).

Combining equations 7.1 and 7.2 yields

$$\sum_{i,j}\left[\frac{I_{i,j}}{V_j}\right]d_i^h k^h H_j < D_{\max}^h \tag{7.3}$$

or

$$\sum_{i,j}\left[\frac{I_{i,j}}{V_j}\right]H_j / X_i < 1 \tag{7.4}$$

where

k^h = $k_1 * k_2$, i.e., the factor that accounts for the fraction of waste exhumed during drilling, the mixing of the waste in the soil, and transport to point of exposure (1/m), and

X_i = limiting value of [I/V]*H for contaminant i (Ci/m^2) = $D^h_{max}/(d_i^h k^h)$.

Recognizing that average values were used in the analyses presented in Chapter 5, the values of X_i can be calculated easily from data already presented in this PA.

$$\left[\left(\frac{I_i}{V}\right)H\right]_{\max} = X_i = \frac{D_{\max}^h}{k^d d_{Sn}^h}\left(\frac{I_i}{I_{Sn}}\right)\left(\frac{D_{Sn}^h}{D_i^h}\right) \tag{7.5}$$

where

I_i = the average inventory of contaminant in each container, each container having a volume V and a height H

D^h_i = the intruder dose for contaminant i, and

$$k^h d_{Sn}^h = \frac{D_{Sn}^h}{\sum_j [I_{Sn}/V]H} \tag{7.6}$$

The values for I are obtained from the column labeled "ILAW Package" in Table 3-1 that contains the average inventory in an ILAW package. Intruder dose values (D) are taken from Table 5-2 for the acute (driller) scenario and from Table 5-3 for the continuous (homesteader) scenario. The waste volume assumed in Chapter 5 is a cube 1.4 m on each side filled to 85 percent capacity, which corresponds to a waste height of 1.2 m.

The values for the dimensions of the ILAW package have changed over the past few years along with the number of packages in a vertical column. Therefore, restrictions are placed assuming that the dimensions and number of packages vary.

Table 7-8 displays the restrictions due to protection of the inadvertent intruder.

Table 7-8. Restrictions Based on Protecting Inadvertent Intruder.

$\Sigma[I_{ij} / V_j] H_j / X_i < 1.0$. All waste disposed must meet the NRC Class C limits, which also are treated as a sum of fractions rule. All waste produced during Phase 1 must have a lower average concentration than is shown in the last column.

Isotope	X_i or Max $[\Sigma I_{ij}/V_j]H_j$ (Ci/m^2)	Max $[I_{ij} / V_j]$ (Ci / m^3) $X_i / \Sigma H_j$ (1)	NRC Class C	Avg $[I_{ij} / V_j]$ Phase 1 Contract
^{59}Ni	3.80E+04	6.11E+03	-	-
^{60}Co	2.48E+29	4.00E+28	-	-
^{63}Ni	5.63E+05	9.07E+04	-	-
^{79}Se	1.39E+03	2.23E+02	-	-
^{90}Sr	3.79E+06	6.11E+05	7,000.	20.
^{93}Zr	5.22E+03	8.41E+02	-	-
$^{93}Nb^m$	6.88E+13	1.11E+13	-	-
^{99}Tc	2.18E+02	3.51E+01	3.	0.1
^{106}Ru	3.72E+158	5.98E+157	-	-
$^{113}Cd^m$	3.85E+12	6.19E+11	-	-
^{125}Sb	5.06E+58	8.15E+57	-	-
^{126}Sn	7.34E-02	1.18E-02	-	-
^{129}I	1.90E+01	3.06E+00	0.08	0.08
^{134}Cs	1.95E+77	3.15E+76	-	-
^{137}Cs	6.19E+04	9.97E+03	4,600.	3.
^{151}Sm	5.28E+06	8.51E+05	-	-
^{152}Eu	1.68E+11	2.70E+10	-	-
^{154}Eu	7.37E+17	1.19E+17	-	-
^{155}Eu	1.37E+35	2.21E+34		
^{226}Ra	1.02E-01	1.64E-02	0.027 (2)	0.027 (2)
^{227}Ac	8.47E+06	1.36E+06	0.027 (2)	0.027 (2)
^{228}Ra	5.36E+26	8.64E+25	0.027 (2)	0.027 (2)
^{229}Th	5.01E-01	8.08E-02	0.027 (2)	0.027 (2)

Table 7-8. Restrictions Based on Protecting Inadvertent Intruder.

$\Sigma[I_{ij} / V_j] H_j / X_i < 1.0$. All waste disposed must meet the NRC Class C limits, which also are treated as a sum of fractions rule. All waste produced during Phase 1 must have a lower average concentration than is shown in the last column.

Isotope	X_i or Max $[\Sigma I_{ij}/V_j]H_j$ (Ci/m^2)	Max $[I_{ij} / V_j]$ (Ci / m^3) $X_i / \Sigma H_j$ [1]	NRC Class C	Avg $[I_{ij} / V_j]$ Phase 1 Contract
^{231}Pa	2.86E-01	4.60E-02	0.027 [2]	0.027 [2]
^{232}Th	5.68E-02	9.15E-03	0.027 [2]	0.027 [2]
^{232}U	1.85E+01	2.99E+00	0.027 [2]	0.027 [2]
^{233}U	8.29E+00	1.33E+00	0.027 [2]	0.027 [2]
^{234}U	4.47E+01	7.20E+00	0.027 [2]	0.027 [2]
^{235}U	1.00E+00	1.62E-01	0.027 [2]	0.027 [2]
^{236}U	6.83E+01	1.10E+01	0.027 [2]	0.027 [2]
^{237}Np	6.95E-01	1.12E-01	0.027 [2]	0.027 [2]
^{238}Pu	1.13E+03	1.82E+02	0.027 [2]	0.027 [2]
^{238}U	6.18E+00	9.94E-01	0.027 [2]	0.027 [2]
^{239}Pu	1.51E+01	2.44E+00	0.027 [2]	0.027 [2]
^{240}Pu	1.58E+01	2.55E+00	0.027 [2]	0.027 [2]
^{241}Am	1.86E+01	3.00E+00	0.027 [2]	0.027 [2]
^{241}Pu	5.44E+02	8.76E+01		
^{242}Cm	2.22E+05	3.58E+04	0.027 [2]	0.027 [2]
^{242}Pu	1.57E+01	2.53E+00	0.027 [2]	0.027 [2]
^{243}Am	8.79E-01	1.41E-01	0.027 [2]	0.027 [2]
^{243}Cm	1.11E+04	1.78E+03	0.027 [2]	0.027 [2]
^{244}Cm	5.72E+03	9.21E+02	0.027 [2]	0.027 [2]

[1] A stacking height of 6.2 m (3 packages * 2.3 m each * 90 percent full) is used.
[2] NRC requirements are expressed in curies per gram. A density of 2.7 g/cm^3 was used for the waste form.

7.6.3 Requirement Development Based on Groundwater Protection

There are three performance measures that need to be used when using the groundwater scenario (all-pathways dose, beta/photon drinking water doses, and concentration of alpha emitters). Moreover, contamination of the groundwater could occur due to 1) the waste initially inside the ILAW packages reaching the groundwater, or 2) smearable contamination on the outside of the ILAW packages reaching the ground water.

7.6.3.1 Waste Inside ILAW Package. The long times, the Impact (D^{gw}) from the groundwater scenario (as discussed in Section 4.11) can be simply expressed and this impact must be less than the performance goal (D^{gw}_{max})

$$D^{gw} = \sum_i (I_i R) d_i^{gw} k^{gw} < D_{max}^{gw} \tag{7.7}$$

or

$$\sum_i {I_i R}/{Y_i} < 1 \tag{7.8}$$

where the sum is over all contaminants i and where

I_i = the inventory of contaminant i (Ci)

R_i = the fractional release rate of contaminant i from the waste form (1/y)

d_i^{gw} = the dosimetry factor relating response to concentration of contaminant i in the groundwater scenario [(mrem/year)/(Ci/m^3) or 1 in the case of concentration]

k_i^{gw} = the factor that accounts for vadose zone and aquifer transport (years/m^3 or y/L)

D_{max}^{gw} = the maximum impact (drinking water dose, all-pathways dose, or concentration of alpha emitters) allowable in the groundwater scenario (see Table 7-6) (mrem/year or Ci/L)

Y_i = limiting value for (IR) [Ci/year].

The values of Y_i easily can be calculated from information already presented in this PA.

$$[I_i R]_{\max} = Y_i = \frac{D_{\max}^{gw}}{k_i^{gw} d_{ref}^{gw}} \left(\frac{I_i}{I_{ref}} \right) \left(\frac{D_{ref}^{gw}}{D_i^{gw}} \right) \tag{7.9}$$

where

D^{gw}_i = groundwater impact for contaminant i, and

$$k_i^{gw} d_{ref}^{gw} = \frac{D_{ref}^{gw}}{I_{ref} R_{\max}} \tag{7.10}$$

where ref = Tc for mobile ($K_d = 0$) contaminants and ref = ^{237}Np for retarded contaminants. The values for I are obtained from the Table 3-1 column labeled "ILAW Inventory." That column contains the total ILAW inventory. The impact values (D) are taken from Tables 4-4, 4-5, and 4-6 for the beta-photon drinking water dose, the all-pathways dose, and the concentration of alpha emitters, respectively, for the important radionuclide contributors. The maximum release rate (R_{max}) is 8×10^{-7}/year, which is taken from Table 4-1. (The maximum release rate is the value after about 20,000 years).

Because the details of the separations or treatment process that creates ILAW the waste form have not been determined, neither the ILAW inventory nor the release rate is known. Therefore, the Immobilized Waste Program will place restrictions on the product of the inventory (I_i) and on the release rate (R_i).

In the 1998 ILAW PA (Mann 1998a), an additional term was included in equations 7.7 and 7.8. This term, appearing as a divisor, accounted for different groundwater dilution as the length of the disposal facility perpendicular to the groundwater flow became larger. However, because the groundwater flow in this PA flows in the ancestral Columbia River channel, the disposal facility length is not especially significant. The width of the channel is more responsible for establishing the dilution factor, rather than the dimensions of the disposal facility.

Using these equations and associated values, restrictions on the release rate were calculated and are displayed in Table 7-9.

For groundwater protection, the most restrictive limit based on the beta-photon drinking water dose, alpha concentration, and all-pathways was used. Because the waste form performance is so important, the product of the total inventory times the release rate is important rather than the total inventory for a radionuclide.

As noted in Section 4.8.2, there is a potential for chromium from the ILAW container to be a problem. However, until more is known about the conversion of metallic chromium to Cr(VI) under disposal facility conditions, it is premature to set restrictions on the amount of chromium in the ILAW containers.

Table 7-9. Restrictions Based on Protecting Groundwater - ILAW Package.
$\Sigma[I_{ij} R] / Y_i < 1.0$.

Isotope	Y_i Max IR ($Ci\ y^{-1}$)	Limiting Quantity
^{99}Tc	7.93E-01	Drinking water dose
^{129}I	4.06E-03	Drinking water dose
^{231}Pa	2.87E-03	All pathways dose
^{233}U	3.69E-02	Alpha concentration [1]
^{234}U	3.60E-02	Alpha concentration [1]
^{238}U	3.65E-02	Alpha concentration [1]
^{237}Np	1.20E-02	All pathways dose

[1]Although uranium is not covered under the Federal Drinking Water regulations for alpha emitter concentration, it is included here.

7.6.3.2 Smearable Contamination. Although smearable contamination is not expected on the outside of the ILAW packages, some, even if not a measurable amount, is likely to be present. In this case, the restriction can be interpreted as

$$\sum_i I_i^{smear} d_i^{gw} k^{smear} < D_{\max}^{smear} \tag{7.11}$$

or

$$\sum_i I_i^{smear} / Z_i < 1 \tag{7.12}$$

where the sum is over all contaminants i and where

I_i^{smear} = the inventory of smearable contaminant i (Ci)

d_i^{gw} = the dosimetry factor relating response to concentration of contaminant i in the groundwater scenario [(mrem/year)/(Ci/m^3) or 1 in the case of concentration]

k_i^{smear} = the factor that accounts for vadose zone and aquifer transport (m^{-3} or 1/L)

D^{smear}_{max}= the maximum impact (drinking water dose, all-pathways dose, or concentration of alpha emitters) allowable in the groundwater scenario (taken as values in Table 7-6)(mrem/year or Ci/L)

Z_i = limiting value for I^{smear} (Ci).

The values of Z_i can be calculated easily from information already presented in this PA.

$$\left[I_i^{smear}\right]_{max} = X_i = \frac{D_{max}^{gw}}{k^{smear} d_{ref}^{gw}} \left(\frac{I_i}{I_{ref}}\right)\left(\frac{D_{ref}^{gw}}{D_i^{gw}}\right) \tag{7.13}$$

where

D^{gw}_i = groundwater impact for contaminant i, and

$$k^{smear} d_{ref}^{gw} = \frac{D_{ref}}{I_{ref}} \tag{7.14}$$

where ref = ^{99}Tc for mobile (K_d = 0) contaminants and ref = ^{237}Np for retarded contaminants. The values for I are obtained from the column labeled "ILAW Inventory" in Table 3-1 that contains the total ILAW inventory. The impact values (D) are taken from Tables 4-20, 4-21, and 4-22 for the beta-photon drinking dose, the all-pathways dose, and the concentration of alpha emitters, respectively. The values for D^{gw}_{max} are the same as for the groundwater scenario.

Using these equations and associated values, restrictions on the amount of smearable contamination were calculated and are displayed in Table 7-10.

Table 7-10. Restrictions Based on Protecting Groundwater - Smearable Contamination. [$\Sigma I_i / Z_i < 1.0$].

Isotope	Limiting Quantity	Z_i Max I^{smear} (Ci)	Max I^{smear} (pCi/m^2) (2)
^{99}Tc	Drinking water dose	5.91E+02	1.28E+09
^{129}I	Drinking water dose	3.11E+00	6.71E+06
^{231}Pa	All pathways dose	1.64E+00	3.53E+06
^{233}U	Alpha concentration [1]	2.26E+01	4.88E+07
^{234}U	Alpha concentration [1]	2.22E+01	4.78E+07
^{238}U	Alpha concentration [1]	2.23E+01	4.80E+07
^{237}Np	All pathways dose	7.31E+00	1.58E+07

[1]Because uranium is not covered under the Federal Drinking Water guidelines, the all-pathways dose is used as limiting quantity.

[2]Total area of waste packages taken as 463,500 square meters.

7.6.4 Waste Form Restrictions

Using the information generated in Sections 7.6.2 and 7.6.3, restrictions based on long-term performance can be placed on the ILAW glass and package. These fall into three categories, restriction on the waste concentration, waste release rate, and smearable contaminants on the surface of the ILAW package.

7.6.4.1 Concentration. The restrictions on waste concentration due to long-term performance are displayed in Table 7-8 (second column) and depend upon not only the waste concentration, but also the ILAW package height and the number of layers used in the disposal facility. However, using the current plans for package height and disposal facility operation [containers being 1.22 m in diameter by 2.3 m high, filled with 90 percent glass (DOE/ORP 2000c) and being stacked three containers high], concentration limits were derived and are displayed in the third column of Table 7-8. These limits are compared to the NRC class limits (fourth column of Table 7-8) and the average concentration presently in the contract (fifth column).

In general, the NRC Class C requirements are the most restrictive. For those radioisotopes having Class C restrictions, the limits from this analysis are more restrictive for ^{137}Cs, ^{231}Pa, ^{232}Th, and ^{235}U. For the non Class C radioisotopes, the most important restriction is for ^{126}Sn. However, for ^{90}Sr, ^{99}Tc, and ^{129}I, the Phase 1 contract is much more restrictive than either the limitations calculated here or NRC Class C limits.

7.6.4.2 Release Rate. For most risk analyses, the release time is short compared to travel time, and hence the release rate is unimportant. In these cases, the total inventory is restricted by the risk analysis. For ILAW disposal, however, the release time is much longer than the travel time. Thus, it is the product of the inventory and the release rate that is important. The restrictions on the product of inventory and release rate are shown in Table 7-9. Although it is the product that is important, using the current inventory, the contaminant release rate from the waste form should be less than 137 ppm/year.

7.6.4.3 Smearable Contamination.

The limits for smearable contamination are quite high and are displayed in Table 7-10. Less than 0.1 percent of the inventory of the key contaminants can appear on exposed surfaces as smearable contamination.

7.6.5 Disposal Facility Requirements

Most of the requirements imposed by the performance assessment analysis are on the waste form. However, a few are imposed on the disposal facilities. The major facility requirements deal with subsidence, recharge rate, layout, interactions with the waste form, and intruder protection.

The performance assessment assumes that subsidence is small based on the slow degradation of the waste form and the use of filler materials to minimize voids in the disposal

facility. This means that the facility must be constructed without significant void space. In addition, after waste is placed inside the facility, the spaces between the waste containers must be filled with a dry material. The estimated impacts of a 1 -m wide subsidence of 0.3 m depth in the remote-handled waste trench facility located 10 m from the RCRA-compliant cap apex and extending the long length of the trench are factors of 500 and 1,000 times larger for the alpha concentration and beta-photon drinking water dose, respectively, than the estimated impacts for the best estimate case at 10,000 years after facility closure. Facility performance enhancements associated with the RCRA-compliant cap and the capillary break can be effectively lost if massive subsidence occurs. For the subsidence case analyzed , the estimated impacts still are less than the estimated impacts for the base analysis case.

Because the waste form releases contaminants at a slow, approximately steady rate, the time dependence curve for exposure is shaped more like a plateau than a peak. Therefore, a major effect of the recharge rate is to affect the arrival of contaminants to the groundwater. If the second group of contaminants (i.e., those having $K_d = 0.6$ mL/g, such as uranium) arrived in significant quantities before 10,000 years, the all-pathways dose would greatly increase and restrictions may have to be placed on the recharge rate. The base analysis case shows that achieving the natural recharge rate (4.2 mm/year) is sufficient to meet performance objectives. If a subsurface sand-gravel capillary barrier is used, the infiltration rate could be far lower.

The requirement for groundwater protection [$\Sigma(I_i R_i) / Y_i < 1$] actually is on the disposal system. The designers of the disposal structures must ensure that materials are not used that would accelerate waste form degradation. Alternatively, the designers can add components (for example, hydraulic diverters, getters) to minimize the requirements on the waste form.

Designers of the engineered system may wish to add components to provide greater defense in depth. The major components would be an improved surface barrier to reduce the recharge rate, a hydraulic barrier to divert moisture from the waste, concrete pads to trap uranium, and other getter materials to trap important radionuclides such as technetium. The recharge rate is the main driving function for the system. Having a surface barrier that could reduce this rate would lengthen the time the contaminants take to reach the groundwater. Diverting water from the waste would likely reduce the contaminant release rate from the waste form and create a greater moisture shadow under the disposal system that also would delay contaminant travel. Concrete is known to highly retard movement of uranium isotopes, thus reducing their impact during the time of compliance. If an inexpensive getter could be found for technetium, such a material could also have important impacts.

7.6.6 Likelihood of Meeting Requirements

The likelihood of meeting the requirements for the waste form and the disposal facility is determined by establishing a case using the best available information.

Because the base analysis case met the performance objectives and the sensitivity cases showed the robustness of the assumptions and data, this system can reasonably be expected to meet the

requirements imposed in Sections 7.6.3 and 7.6.4. This section demonstrates that compliance.

7.6.6.1 Inadvertent Intruder Compliance Case. Current plans have a different container size and number of containers per stack than the configuration analyzed in Chapter 5. However, these plans are reflected in the requirements given in Table 7-8.

Table 7-11 compares the average inventory concentration and the bounding concentration in an ILAW package to the inadvertent intruder protection limits from Table 7-8. As expected from the results described in Chapter 5, in most cases the expected and bounding inventories are much less than allowed. The exceptions are some actinides (^{239}Pu, ^{241}Am, and ^{241}Pu), where the inventory must be reduced to meet Class C limits as required by the contract to produce ILAW (DOE/ORP 2000c).

Table 7-11. Comparison of ILAW Inventory Concentrations to Allowed Limits. (NRC Class C Limits Have Not Been Applied.)

Nuclide	Ratio of Averaged Inventory/ Allowed Inventory*	Ratio of Bounding Inventory /Allowed Inventory*
90-Sr	0.004	0.008
99-Tc	0.012	0.033
126-Sn	0.091	0.880
232-Th	0.001	0.007
232-U	0.008	0.061
233-U	0.031	0.230
234-U	0.010	0.072
237-Np	0.019	0.066
238-Pu	0.025	0.100
238-U	0.011	0.075
239-Pu	0.715	3.519
240-Pu	0.123	0.496
241-Am	2.537	62.393
241-Pu	1.678	7.333
242-Cm	0.013	0.430
243-Cm	0.002	0.019
244-Cm	0.024	0.251

*Allowed inventory is not corrected for decay

7.6.6.2 Groundwater Protection Compliance Case. For groundwater, the results do not change between the base analysis case and the compliance case. Thus the comparisons given in Section 7.2 are still valid. That is, a large margin exists between the requirements and the expected performance.

7.6.6.3 Smearable Contamination Compliance Case. The limits for smearable contamination (Table 7-10) are high. The contract specifications are that no more than 367 Bq/m^2 (or 1.22×10^4 pCi/m^2) of alpha contamination and no more than 3670 Bq/m^2 (or 1.22×10^5 pCi/m^2) of beta-gamma contamination should be present on the surface. Comparing the requirements of Table 7-10 to these numbers shows that this performance assessment is much less restrictive than the contract, especially with the safety margin built into this analysis.

7.7 SUMMARY OF THE IMPACT OF DIFFERENCES BETWEEN THE 1998 ILAW PA AND THIS DOCUMENT

Of the three types of scenarios (groundwater, air, and inadvertent intruder) studied in the 1998 ILAW PA (Mann 1998a) and this document, only the results for the groundwater scenario are significantly different. Five major differences occur in inputs between the 1998 ILAW PA and this document that affect the peak values of impact parameters for scenarios that contaminate groundwater:

- Time of compliance
- Inventory of mobile constituents
- Disposal facility design
- Waste form performance
- Groundwater dilution.

Other new data (such as recharge rates, geochemistry, and hydrology) affect the time that the peak occurs or the impact parameters through one of the last four inputs cited.

The 1998 ILAW PA used 10,000 years as the time of compliance. Because of new DOE guidance, the present time of regulatory compliance is 1,000 years. However, because of the slow travel time in the vadose zone, even the mobile constituents do not reach the groundwater in any significant quantity in only 1,000 years. To make comparisons with the 1998 ILAW PA easier, Table 7-12 summarizes the differences in impact parameters at 10,000 years.

The facility design effect is associated with areal distribution of the waste. For the remote-handled waste trench disposal concept, the areal footprint for the facility is 124,800 m^2. For the 1998 ILAW PA, the Concept 1 disposal facility had an areal footprint of 51,000 m^2. The larger areal distribution of the waste leads to a reduction factor of 0.41 associated with the contaminant concentration entering the aquifer over a larger area.

Table 7-12. Effect of Updated Model Inputs on the Estimated Beta/Gamma Drinking Water Dose at 10,000 Years. (1998 ILAW PA estimated this dose as 2.0 mrem/year.)

Updated Model Input	Beta-Gamma Drinking Water Dose
	Ratio 2001 ILAW PA to 1998 ILAW PA
Facility design	0.41
Technetium inventory	0.26
Ratio of other mobile contaminants to Tc[1]	1.32
Technetium dose factor	0.83
Waste form release rate/ vadose zone transport	0.30
Groundwater dilution	0.14
All inputs	0.0049

[1]based on updated K_d values for selenium, iodine, and neptunium.

The estimated impact at 10,000 years of changing the inventory of the mobile constituents is to reduce the present estimated groundwater impacts by a factor of 0.34 (0.26 * 1.32) when compared to the results from the 1998 ILAW PA (Mann 1998a). The lower estimated impact of inventory in the present analysis arises from two changes: the change in the inventories of ^{99}Tc, the most important radionuclide in either analysis, and of other mobile radionuclides. The 1998 ILAW PA assumed that 80 percent of the technetium in the tanks would end up in ILAW, while this document assumes, based on the contract between the treatment vendor and DOE, that only 20 percent of the technetium in the tanks will go into ILAW. The remaining slight difference in the technetium inventory results from a small change in tank inventory.

Based on disposal site-specific geochemical measurements, the determination of which contaminants are mobile has changed somewhat. Technetium-99 still is the most important mobile contaminant. In the 1998 ILAW PA, ^{79}Se was assumed to be mobile because no Hanford Site-specific data were available that indicated otherwise. Since then, we have learned that the half-life of ^{79}Se is longer than was believed. Also, disposal-site specific information has shown that selenium transport in the vadose zone is chemically retarded. However, iodine and neptunium, which were treated as relatively immobile in the 1998 ILAW PA, now are known through disposal-site specific information to be more mobile. Thus, where ^{99}Tc was 75 percent of the drinking water dose in the 1998 ILAW PA, it is only 57 percent in this document. Therefore, the relative contribution from other mobile contaminants has increased to 1.32 (0.75/0.57).

The new DOE O 435.1 requires the use of the EPA dose factors; the dose factor for ^{99}Tc was 0.83 of the dose factor used in the 1998 ILAW PA. In the 1998 ILAW PA, the release from the vaults was assumed to be that given in the request for proposal for treatment services (4.0×10^{-6}/year) (DOE/RL 1996). At 10,000 years after facility closure, the contaminant flux to the aquifer was 2.0×10^{-6}/year. In this document, the release from the

remote-handled waste trench is calculated by simulating the waste form release (rate = 0.8×10^{-6}/year at 10,000 years after facility closure) from LAWABP1 glass and performing the transport of contaminants through the vadose zone resulting in a contaminant flux of 0.7×10^{-6}/year at 10,000 years after facility closure. This results in a 30 percent decrease in the contaminant flux to the aquifer when compared to the 1998 ILAW PA values.

We now realize that the disposal site is situated over the old channel of the Columbia River. Also, the base analysis case used a recharge rate of 3 mm/year in the 1998 ILAW PA and a rate of 4.2 mm/year in this analysis. The hydraulic conductivity of the unconfined aquifer is higher, resulting in greater dilution, by about a factor of 7.

Combining these factors (inventory of mobile constituents, disposal facility design, waste form performance, and groundwater dilution), the overall effect is a reduction by a factor of about 200 from the 1998 ILAW PA for the estimated impacts to the groundwater.

7.8 FURTHER WORK

The ILAW performance assessment team is collecting information for the next version of this document. Plans for data collection and documentation for the program's life have been issued.

The Immobilized Waste Program is committed to providing information to the DOE and the public on the long-range human health and safety and the environmental impacts from the disposal of immobilized low-activity waste. The program realizes that this effort is just the beginning. The program realizes that although the analysis presented in this performance assessment is robust, it relies on data that may not reflect the actual facility design or the composition of the waste being disposed. The maintenance of the ILAW performance assessment is documented in the *Maintenance Plan for the ILAW Performance Assessment* (DOE/ORP 2000), which was approved by the DOE/ORP field manager and sent to DOE/LFRG as required (French 2000b).

The Program will provide additional analyses based on new data and methods as they become available so that the decision makers can make the appropriate decisions. To support these assessments, an extensive data collection and interfacing effort is planned (Puigh 2000). The type and amount of new data will be guided by the uncertainty of current data and the importance of such data to the results of future analyses. The major components of this data collection effort are as follows:

- Waste form data collection and model development
- Interactions with disposal facility designers
- Geotechnical data collection and model development
- ILAW inventory
- Other data collection (dosimetry, scenario development, performance objective selection).

7.8.1 Future Performance Assessments

As noted in Section 1.4 and presented in Table 1-2, a number of PAs will be produced that incorporate the latest data and information and that present the long-range impacts from disposal. These future PAs are expected to reaffirm the conclusions reached in this document. The next PA is planned to be submitted in 2005 to support the start-up of disposal operations. After that, PAs are scheduled on 5-year cycles. These later performance assessments would support new designs, the startup of new facilities, and the closure of filled facilities.

As with the 1998 ILAW PA and this document, a set of data packages will be published that will contain all data to be used in the 2005 ILAWPA. Each data package will contain the best estimate of the information needed for the analysis (and its uncertainties), along with reasonably bounding information so that the analysis can state what will not happen. Each data package will be reviewed; in the more important cases, the review will be performed by outside experts. If new data not contained in the data packages are used in the PA, they will be carefully documented and justified in the PA itself. The performance assessment will carefully follow the guidance provided by the DOE and information from the technical reviews of other PA documents and Hanford Site-relevant environmental analyses.

7.8.2 Waste Form Data Collection and Model Development

This analysis has shown that waste form behavior is the key parameter to the disposal of immobilized low-activity tank waste. Because of the projected slow release, the engineered system rather than the natural system will mainly determine the peak exposures.

In support of the 1998 ILAW PA, a significant effort was spent characterizing the low-level waste glass LD6-5412. Based on that experience, a plan was developed for estimating waste form performance thousands of years from now. Since then, a tremendous amount of work has been performed on a variety of potential low-level waste glasses, particularly LAWABP1. Based on this experience and on the review of that plan (SRC 2001), the plan remains basically unchanged and is presented in Appendix E. The plan still forms the core of the performance assessment waste form task.

Estimating long-term waste form performance will be based on simulating the environment and waste form as a function of time and position in the disposal facility. To be successful, such a strategy depends on an understanding of the processes involved, on a powerful computer code that incorporates those processes, and on well-tested and -understood parameters for those processes.

As noted in Section 3.3.3.2, scientists believe they understand the major processes of glass dissolution and contaminant release. The release of contaminants from a glass waste form in a dry environment over thousands of years is complex. Improvements on codes such as STORM are expected. The improvements will allow the code to better incorporate not only the chemical effects of waste form release, but also the hydraulic principles that govern the contaminant transport in the facility.

Four major types of experiments (SPFT, VHT, PCT, and PUF) will be used. The samples in the tests will be produced using the proposed formulations of the ILAW producers and radioactive waste from Hanford Site tanks. These tests not only will provide parameters for the computer code analyses, but will serve to validate the codes. Initially, tests will be performed with nonradioactive simulants of tank waste. Later tests on glasses made with actual waste will be performed.

The single pass flow-through (SPFT) tests determine how the glass dissolves under constant environmental conditions (temperature, pH, ion concentration). These tests pass a water solution of known ionic concentration at known temperatures through the glass. Because the dissolved glass never forms a significant fraction of the fluid, parameters such as the pH power law coefficient, the activation energy, and the rate law order can be determined. Also by investigating the entire data set, possible deviations from the model can be investigated and understood.

The vapor hydration tests (VHT) allow the determination of the formation of phases from the dissolved glass materials. Such phases could accelerate glass dissolution or could trap contaminants in a stable medium. These measurements normally are done at temperatures higher than 100 °C, which greatly accelerates glass dissolution and formation of the phases. By performing the tests at different temperatures, scientists can investigate the temperature effects and extrapolate the measurements to temperatures typical of ILAW disposal. VHT also can supplement the results of the product consistency test (PCT), which is run at lower temperatures.

PCT is a well-established test to determine how glass will dissolve in a static environment. In this test, the environmental conditions are established at the beginning of the test, then measured as the test proceeds. Because the ILAW waste contains so much sodium, the environmental conditions (particularly pH) change significantly. PCT tests are run at a variety of temperatures and surface-to-volume ratios. Such measurements allow the pseudo-equilibrium constant to be determined. These tests not only provide an integral check on the SPFT results, but also can provide information on the importance of reactions such as the sodium ion exchange reaction.

The pressurized unsaturated flow (PUF) test is unique among the tests because it is the only test performed in an unsaturated-moisture environment. In this test, water is forced to flow through a porous medium of glass at a slow rate. Because the disposal facility is expected to be extremely dry, the PUF tests will be used to determine whether (or how) any of the parameters or processes change under such dry conditions. PUF test results already have been used to determine the change in hydraulic conductivity as the glass dissolves, as well as determining other physical properties. The PUF tests, particularly those carried out over long periods, are expected to severely test the models and our understanding of them.

All this test information is being used to formulate, test, and calibrate the computer models, which are being modified to make them more robust and accurate. More chemical processes are being modeled. Our intent is to have a computer model that simulates the influx of water to the disposal facility, the corrosion of the glass, the release of the

contaminants from the original glass waste form and its secondary mineral phases, and the transport of those contaminants out of the engineered disposal system.

7.8.3 Interactions with Disposal Facility Designers

Disposal facility construction costs will be significant. The performance assessment team has been and will continue to work with the disposal facility designers to identify key components of the facility design so that cost-effective, safety-enhancing decisions can be made. A prime example of this was the decision to use a disposal facility design based on trenches instead of concrete vaults. Not only is the trench design more cost effective, but also contaminants will be released at a slower rate from it, causing less environmental impact. The performance assessment team will continue to work closely with the design team to determine the components that must be included in the models for future performance assessments.

To model the disposal facility, we need not only the design, but also the hydraulic and transport properties of the waste form and near-field environment materials. Movement of moisture through cracks in the waste form and the hydraulic and geochemical properties of the original and degraded materials will be investigated. Efforts to support design of surface and sand-gravel capillary moisture barriers, getter materials, and water conditioning layers are part of the multi-year plan.

7.8.4 Geotechnical Data Collection and Model Development

Geotechnical data describe the vadose zone and the unconfined aquifer. The vadose zone and the unconfined aquifer play an important role in the performance assessment because the vadose zone delays the arrival of contaminants and the aquifer supplies additional water with which the contaminants can mix. An integrated program covering all aspects of the Hanford Site's vadose zone and unconfined aquifer has just begun. The program should result in more efficient data collection and a more thorough method review.

In each of the next 2 years, a borehole will be drilled at the disposal site. These boreholes will provide samples for determining other geotechnical information. They also will allow access to the vadose zone for in situ moisture experiments and to the unconfined aquifer for groundwater testing. The samples from the boreholes, along with samples from other locations, will be used to determine the geologic strata underlying the disposal facilities and the hydraulic and geochemical properties of those strata. In addition, the samples will be used to determine chlorine content as a function of depth, which will provide the recharge rate over the last 13,000 years).

Other analyses will support the determination of geotechnical data needed and use those data once they are gathered. For example, the computer simulations of moisture flow in Hanford Site surface sediments will combine the long-term time-varying infiltration rates determined by chlorine measurements (both total Cl and ^{36}Cl) with short-term (a few decades) determination of the infiltration rate and climate, soil, and vegetation studies. Enhanced contaminant transport through colloidal movement or in clastic dikes also will be

studied. Studies of the spatial variability of soil hydraulics data and alternative conceptual models will be important for completing future performance assessments.

Because of the thickness of the vadose zone and the size of the disposal facilities complex geotechnical computer codes and models are needed to forecast future moisture flow and contaminant transport. These codes and models will be improved as new data and methods are developed.

7.8.5 ILAW Inventory

The inventory in each ILAW package will depend on the radionuclide inventories of the 177 individual tanks, the order and method of retrieval from those tanks, the blending on the retrieved contents, and the separations and the immobilization processes used by the ILAW producers.

The PA team will continue its close interface with the RPP Standard Inventory effort. The team also will closely interact with the RPP Retrieval Group to incorporate their plans into the ILAW estimates. Finally, contacts with the ILAW producers will occur to determine their separations and immobilization plans to better estimate the fraction of the waste they receive that will end up in low-activity tank waste product. The Program also will use such estimates for safety studies.

Once the ILAW packages are produced, the PA team will rely on product manifests. Thus, as ILAW packages are produced and product manifests become available, the complexity and uncertainty of the inventory estimate is expected to decrease.

7.8.6 Other Data Collection

Other information must be obtained for a successful PA. Dosimetry data must be collected. Scenarios must be developed and reviewed. Performance objectives must be selected and reviewed.

The Hanford Site has a DOE-chartered group (the Hanford Environmental Dose Oversight Panel) to standardize dosimetry data and methods and to review the results of calculations. The PA team will work closely with this group to ensure that its data and methods comply with Site standards.

The public has strong views concerning scenarios and performance objectives. The PA team will review existing guidance, Hanford Site activities (especially the environmental impact statement activities), and other Hanford Site project efforts, such as the Hanford Groundwater / Vadose Zone Integration Project's regulatory path forward effort, to determine the scenarios and performance objectives to be used in future PAs.

7.9 CONCLUSION

This performance assessment analyzed the long-term environmental and human health impact of disposing of immobilized low-activity waste from Hanford Site tanks. This analysis confirms the conclusions of the 1998 ILAW PA that an understanding of ILAW contaminant transport exists and that the proposed disposal action can meet the performance objectives. Based on this expectation, requirements for waste acceptance and disposal facility performance were established. The final analysis of this performance assessment shows a "reasonable expectation" that these requirements will be met. Finally, this chapter concludes by explaining the additional effort that is under way to confirm the data used in this assessment and to document the effects of new information and understanding on the long-term impact of the disposal of this waste.

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Dr. Bacon has over 15 years of experience in vadose zone hydrology and geochemistry and numerical simulations of subsurface flow and transport. As a part of her doctoral studies, Dr. Bacon developed an inverse multiphase reactive transport code to simulate the effect of heterogeneous soil properties and transient changes in transport parameters on fractionation between CO_2 and the source of CO_2 in the vadose zone. In support of the Hanford Tank Low-Activity Waste Glass Performance Assessment Project and the Yucca Mountain Project, Dr. Bacon currently is developing the coupled nonisothermal, multiphase flow and transport code, STORM. She performed the waste form release calculations.

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Mr. Bergeron joined Battelle, Pacific Northwest Laboratory, in May 1985 as a research hydrogeologist and has 21 years of experience in a wide variety of groundwater investigations and studies at hazardous waste and contaminated groundwater sites. Since joining Pacific Northwest Laboratory, Mr. Bergeron has devoted his attention to a variety of roles in hydrologic studies and investigations including technical contributions, project and task management, and line management roles. Technically, he has specialized in the area of hydrogeologic investigations with specific emphasis on applying groundwater flow and transport modeling. He performed the regional- and local-scale groundwater flow and transport modeling used to support this assessment. He led and oversaw the groundwater calculations.

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Before joining DOE, Dr. Brown was a postdoctoral fellow at the Argonne National Laboratory where he studied corrosion characteristics of waste glass and soil remediation. At DOE, he has overseen technology development for the Low-Level Waste Program, as well as the development of TWRS privatization specifications. Currently Dr. Brown is the DOE Contract Officer for the treatment and vitrification effort as well as the DOE manager for the storage and disposal of the immobilized waste. He reviewed this document.

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Mr. Burgard has more than 30 years of experience in engineering, construction, and operations for various mining, processing, and nuclear projects as well as high-level and low-level storage and disposal projects. Relevant PA experience includes program management of the grout PA, project management of the LLW Grout Vault facility construction, program manager for the LLW Technology Group, manager of the ILAW glass testing program, and manager of the IHLW and ILAW conceptual designs to support privatization. Mr. Burgard also was responsible for the construction of numerous coal processing and shipping facilities throughout the United States. He managed this effort and reviewed the document.

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Mr. Freeman has 10 years of experience in the field of hydrogeology. He has experience as a field hydrogeologist and numerical modeler. Mr. Freeman has performed work for both saturated and unsaturated systems and has spent the past 8 years analyzing and modeling moisture and contaminant distribution within the unsaturated sediments beneath the Hanford Site. He performed the vadose zone modeling outside of the waste package area.

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B.S.	Civil Engineering, Humboldt State College	1970
Ph.D.	Engineering (Hydraulics), Utah State University	1979

Dr. Kincaid was the technical manager and key contributor to the *Performance Assessment of the Grouted Double-Shell Tank Waste Disposal at Hanford* and the *Composite Analysis for the Low-Level Waste Disposal in the 200 Area Plateau of the Hanford Site*. He has been the technical group leader of the Soil Physics Group and the Subsurface Transport Group. He was a key contributor in the development of contaminant transport codes and has contributed to various PAs. Dr. Kincaid reviewed this document.

PHILIP E. LaMONT, ~~General Engineer, U.S.~~ Department of Energy Office of River Protection.

B.S.	Chemical Engineering, Washington State University	1965
M.S.	Chemical Engineering, University of Idaho	1975

Mr. LaMont is a project director in the Hanford Site Tank Waste Remediation System Office. He has more than 20 years of experience in nuclear waste management. Mr. LaMont reviewed this document as part of his responsibility for managing projects for storage and disposal of the immobilized waste products from the planned treatment of Hanford Site tank waste.

FREDERICK M. MANN, Scientist, CH2M Hill Hanford Group

B.S.	Physics, Stanford University	1970
Ph.D.	Physics, California Institute of Technology	1975

Dr. Mann is the team leader for the ILAW Performance Assessment activity, which is charged with preparing this document. He was the main author of the 1998 version of the Hanford *Immobilized Low-Activity Tank Waste Performance Assessment*. He has worked for over 25 years in the field of nuclear data and the application of those data to large energy facilities. He has advised the DOE and the International Atomic Energy Agency. He is one of the main authors of this document.

B. PETER MCGRAIL, Staff Scientist V, Applied Geology and Geochemistry Department, Environmental Technology Division, Pacific Northwest National Laboratory.

B.S.	Nuclear Engineering, University of Missouri	1981
M.S.	Nuclear Engineering, University of Missouri	1983
Ph.D.	Environmental Engineering, Columbia Southern University (Magna Cum Laude)	1996

Dr. McGrail has been a staff member at Pacific Northwest National Laboratory for over 17 years and is the principal investigator and manager of a diverse range of projects including the DOE's Fissile Materials Disposition Program, Yucca Mountain Project, Hanford Low-Activity Tank Waste Disposal Project, a basic science (EMSP) project exploring ion-exchange processes in glasses, and a laboratory-directed research and development program exploring carbon sequestration processes in deep geologic formations. He is internationally known for his research on PAs of waste disposal systems, coupled diffusion-advection-reaction transport modeling, and corrosion of silicate glasses, with one patent and more than 150 publications and presentations on these topics. He has performed hundreds of experiments to evaluate the dissolution kinetics of waste forms and other geologic materials using batch and flow-through techniques, and has invented test equipment for conducting these measurements that operates under both water-saturated and unsaturated conditions. He recently completed development of an inverse reactive transport simulator (INVERTS), which is being used to model elution profiles of solutes and colloids from unsaturated flow experiments, including equilibrium and nonlinear kinetic adsorption processes. He also manages the Pacific Northwest National Laboratory's new X-ray Microtomography Laboratory and has been exploring application of high-resolution x-ray microtomography to develop a better understanding of multiphase flow and mass transport processes in porous media. Dr. McGrail directed the collection and analysis of the laboratory data used in the waste form release calculations discussed in this document.

RAYMOND J. PUIGH II, Manager, Nuclear and Environmental Initiatives, Fluor Federal Services.

B.S.	Physics, Louisiana State University	1970
Ph.D.	Physics, Florida State University	1976

Dr. Puigh has over 20 years of experience in nuclear fields ranging from nuclear data and testing to the impact of nuclear radiation on material properties. Relevant performance assessment experience includes metallurgy, tank waste inventory support, safety assessment support for selected tank farm operation projects, and management of environmental modeling staff responsible for the Hanford Solid Waste and ILAW performance assessments. He is one of the main authors of this document.

MARCUS I. WOOD, Principal Scientist, Waste Management, Fluor Hanford, Inc.

B.S.	Geology, University of North Carolina	1973
Ph.D.	Geology, Brown University	1980

Dr. Wood currently is responsible for developing the PA analyses for the disposal of solid low-level waste at the Hanford Site. He is the coordinating author of the Hanford Site solid waste performance assessments and has been largely responsible for the integration and the interpretation of the analytical results in those documents. He has coordinated similar analyses for the Environmental Restoration Disposal Facility (ERDF), at which wastes generated in the remediation of Hanford Site waste sites regulated under the *Comprehensive Environmental Resource Conservation and Recovery Act of 1981* and the 200 West Area low-level burial grounds are disposed. He has directed numerous projects to quantify the geochemical properties of radionuclides in the Hanford Site geohydrologic environment. He also was responsible for developing a multifunctional waste package backfill material for isolating spent fuel and high-level waste. He reviewed this document.

SIGNE K. WURSTNER, Senior Research Scientist, Hydrology Group, Pacific Northwest National Laboratory.

B.S.	Geology, Indiana University	1986
M.S.	Geology, Indiana University	1989

Ms. Wurstner joined Battelle as a full-time scientist October 1989. Her primary interests include numerical modeling of groundwater flow and transport processes and geographic information systems (GIS). Ms. Wurstner has been primarily involved in modeling groundwater flow and contaminant transport at the Hanford Site in both two and three dimensions, and was a major contributor to the development of an interface between the CFEST groundwater modeling software library and the commercial GIS package, ARC/INFO. She assisted in the regional- and local-scale groundwater flow and transport modeling used to support this assessment.

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[2. Section 61.55 describes waste classification. (May 1989)]

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[1. Table A-1 summarizes the amount of waste in the tanks.]

[2. Appendix A provides the status of the tanks.]

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[1. Page 9 discusses use the central plateau wisely for waste management.]

[2. Page 23 discusses the future use options of the 200 Areas.]

[3. Page 25 discusses the cleanup scenario.]

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[1. Section 1.5.1, page 1.15 explains the reason for the selection of 10,000 years as the general time of compliance and of 500 years as the time of inadvertent intruder compliance.]

[2. Sections 3.4.12 on page 3.69 and 3.4.13 starting on page 3.70 describe the vadose zone hydrologic parameters.]

[3. Section 3.3.6.6 on page 3.62 describes the effective diffusion constant for the vadose zone.]

[4. Section 2.6 describes the Grout Disposal Facility.]

[5. The population along the Columbia River is found as footnote f to Table 4-2 on page 4.12.]

[6. Appendix G.1 describes code selection.]

[7. Appendix H describes the benchmarking of PORFLOW 2.394gr against TRACR3D.]

[8. The footnote to Table 3-20 uses a screen depth of 4.6 m.]

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[1. Section 1.6 discusses the time of compliance.]
[2. P. 3-31 provides the discussion of the radon diffusivity coefficient.]
[3. Section 4.3.1 provides the calculation of the release of contaminants to air]

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Appendix A

Comments from
the Low-level Waste Disposal Facility Federal Review Group
(LFRG)
On the 1998 ILAW PA

(Table C-2 of DOE 1999c,
modified to show response for this PA)

LFRG Comments on Performance Assessment for Hanford Immobilized Low-Activity Tank Waste, 7/27/99.				
Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.1 PA is Complete				
3.1.1.a. PA identifies the performance measures used in the PA and a justification of those performance measures as site-specific applications of the performance objectives.	Yes	Performance Measures are defined and reasonably justified in Section 1.6. The performance objective of 25 mrem/year was adopted as a performance measure. The location for compliance is the point of maximum exposure outside a 100-meter buffer zone surrounding the waste. The requirement to assess exposure to a hypothetical inadvertent intruder used the performance measures of 100 mrem/year for chronic exposure and 500 mrem for acute exposure. The time for compliance with these performance measures was taken to be 500 years after facility closure. This time, rather than the default time of 100 years, was justified on the basis of passive barriers and markers. The performance objective to protect water resources was interpreted to require protection of groundwater and surface water. For groundwater protection, the performance measure adopted was that concentrations of contaminants in groundwater not exceed Federal standards for drinking water. Thus, a dose of 4 mrem/year for beta/photon emitters and a concentration of 15 pCi/L for alpha emitters (including uranium) were used. These performance measures were applied to a hypothetical well located 100-meters down gradient from the disposal facility. For surface water, a performance measure of 1 mrem/year was adopted to be consistent with Washington State requirements. The point of compliance for surface water protection was assumed to be the point at which groundwater enters the Columbia river. The performance objective to restrict exposure of any member of the public to no more than 10 mrem/year via the air pathway was adopted as a performance measure. The performance objective to restrict emissions of radon to no more than 20 pCi/m^{2}s was also adopted as a performance measure.	No issues.	Same approach, see Section 1.6.

LFRG Comments on Performance Assessment for Hanford Immobilized Low-Activity Tank Waste, 7/27/99.

Criteria	Criteria met?	Response	Issues	This Document
3.1.1.b. PA presents information on the site geography, demography, land use plans, meteorology, ecology, geology, seismology, volcanology, surface water and groundwater hydrology, geochemistry, geologic resources, water resources, and natural background radiation sufficient to support the analysis presented in the performance assessment.	Yes	The PA presents information on the site geography, demography, land use plans, meteorology, ecology, and regional geology, geochemistry, seismology, volcanology, surface water and ground water resources, and natural background to the reasonable extent necessary. The general discussion on the near-field subsurface geology also seems reasonable; however more details should be presented in the PA. Borehole logs and well completion drawings of the three boreholes (Fig. 2-13) should present more details of the material types, geology, and subsurface hydrology. Tables should be included that list pertinent boreholes and monitor wells with total depth, top of casing, etc. Figure 2-10 poorly represents borehole locations. Consider using: a fold-out plate map, identify latitude and longitude coordinate system, and fence diagram to illustrate geology in area of disposal.	Need better traceability for many details and basic summaries of reference material. Note: Hanford disagreed with this comment based on the need for textual economy.	Figure 2-10 includes fence diagram for ILAW disposal site. New Figure 2-9 shows location of boreholes on ILAW site. Additional data on material types, geology, and subsurface hydrology provided in data packages.
3.1.1.c. PA presents information on the facility design features including elements of the design that address water infiltration disposal unit cover integrity, structural stability, and the inadvertent intruder barrier sufficient to support the analysis presented in the PA.	Yes	Section 2.4 describes the disposal technology. This section includes details of disposal vault construction, both for the existing vaults and for the additional vaults that will be constructed. Sections 2.4.1.4 and 2.4.1.5 discuss closure of the disposal units and the site. These sections discuss features of the closure designed to limit infiltration and those features that deter intrusion. The information presented is sufficient to support the analyses. However, the depiction of the cover design is confusing regarding the slope of the cover. The relationship of cover performance to potential contaminant transport in the vadose zone is not presented clearly. A no-slope cover could create greater deep percolation over the life of disposal. The cover design should consider ALARA concepts (sloping top cover to drain sheet flow in the event of 1000 year precipitation event).	Details regarding cover design are confusing. However, we recognize that conceptual designs were published after issuance of the PA.	Slope of current cover defined in Section 2.4.1. The relationship of cover performance to potential contaminant transport is discussed briefly.

LFRG Comments on Performance Assessment for Hanford Immobilized Low-Activity Tank Waste, 7/27/99.

Criteria	Criteria met?	Response	Issues	This Document
3.1.1.d. PA identifies Federal, state, and local statutes or regulations or agreements that impact site engineering, facility design, facility operations, and the relationship and/or impact of the results of the PA on site engineering, facility design, or facility operations because of these factors.	Yes	In Sections 1.5 and 1.6 all potentially applicable statutes, regulations or agreements are identified. In Section 1.5.2, *the Federal Facility Agreement and Consent Order* is discussed. Some of the legally enforceable milestones cover the Immobilized Waste Program. In Section 1.6.2.2, the *NRC Branch Technical Position on a Performance Assessment Methodology for Low-Level Radioactive Waste Disposal Facilities* is discussed. This document is pertinent because it lays out NRC PA requirements, namely the time of compliance which is 10,000 years. Hanford has adopted the longer time of compliance in order to fully comply with the Incidental Waste requirements of DOE 435.1. In Section 1.6.2.5, Federal and State requirements for drinking water are discussed and applied to the protection of groundwater resources. In Section 1.6.2.6, Federal and State requirements for surface water protection are discussed and applied to the ILAW PA. In Section 1.6.2.7, Federal and State requirements for air emission limits are discussed and applied to the ILAW PA. Section 2.4.1.1 discusses the RCRA design requirement for double containment under which the existing vaults were constructed.	No issues.	Same approach, see Sections 1.5 and 1.6.
3.1.1.e. PA identifies procedures and facility related documentation that may impact site engineering, facility design, or facility operations and the relationship and/or impact of the results of the performance assessment on the documents and site engineering, facility design, or facility operations.	Yes	The PA, in Sections 2.3 and 2.4 identifies documentation (e.g., TWRS record of decision, privatization specifications for immobilization) that may impact the site/facility. However, no evidence is provided that the assumed glass performance can actually be achieved.	No indication is provided that assumed glass performance can be achieved. However, supplemental information provided by Hanford (see Appendix F) mitigated this issue.	The approach for estimating glass performance has changed. It now relies on experiment. See Section 3.4.4

LFRG Comments on Performance Assessment for Hanford Immobilized Low-Activity Tank Waste, 7/27/99.

Criteria	Criteria met?	Response	Issues	This Document
3.1.1.f. PA identifies and justifies key assumptions included in the analysis presented in the PA.	Yes	The PA identifies the key assumptions to be: 1) Inventory assumption that average values from modeling, corrected for credits, seems reasonable and justifiable. 2) Waste form performance assumption that short-term performance and long-term performance are equal is not justifiable without further information. No evidence is presented that the assumed glass performance can actually be achieved. 3) Disposal design, geotechnical considerations, and dose calculation assumptions seem reasonable and justifiable. 4) Recharge relationship to potential contaminant release assumptions needs more explanation in the next iteration of the PA.	1)No issues. 2) Issue is being addressed by Hanford's additional research activities. 3) No issues. 4) Hanford agrees that future Pa revisions will include these effects.	See Sections 1) 3.2 2) 3.4.4 and 7.8.2 3) 2.4, 3.4.3, and 4.3.6 4) 4.5.4
3.1.1.g. PA identifies the point of assessment for each performance measure, and justifies the selection of each point of assessment. *3.1.1.g.1.* The point of assessment for all-pathways, the air pathway excluding radon, and groundwater resource protection is justified based on future land use. If the future site boundary is uncertain, a reasonable point of assessment (e.g., point of maximum impact greater than 100-m from the edge of the disposal unit) is justified.	Yes	The PA, in Section 1.6.2.2 identifies the point of assessment for the all-pathways performance objective as being at the point of maximum exposure, but not less than 100-meters from the disposal facility. The 100-m well is also used for the groundwater protection performance objective as stated in section 1.6.2.5. The point of assessment is conservatively assumed to be the point of maximum exposure outside the 100-m buffer zone for the all pathways, air pathway (excluding radon), and the groundwater protection performance objectives.	No issues.	Same approach, see Section 1.6.

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Criteria	Criteria met?	Response	Issues	This Document
3.1.1.g.2. The default point of assessment for the performance measure for radon exposure that is based on a limit on the average flux of radon of 20 pCi/m^2/s at the ground surface is the ground surface over the disposal unit.	Yes	The performance assessment demonstrates that radon fluxes will be below 20 pCi/m^2/s at the ground surface over the disposal unit. The calculations use appropriate models and assumptions. The deep burial of the waste significantly reduces the small amount of radon present by decay as a consequence of the short half life of radon. The analysis should state that the inventory does not contain sufficient radon precursors to generate significant radon during the time period of analysis. [See Table ES-6; Table 6-5; page 4-74 *et seq.*] Consequently, while the discussion could be improved, the analysis is complete for review purposes even in light of this oversight.	No issues.	Same approach (See Section 4.12).
3.1.1.g.3. The default point of assessment for the alternative performance measure for radon exposure that is based on a limit on air concentration of radioactive material of 0.5 pCi/L is 100-m from the edge of the disposal unit.	Yes	The default performance measure of 20 pCi/m^{2}s is used, thus, this criterion is not applicable	No issues.	Same approach (See Section 1.6.2.7).

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Criteria	Criteria met?	Response	Issues	This Document
3.1.1.h. The performance assessment identifies and quantifies all radionuclides present in the low-level waste to be disposed of at the facility that could significantly contribute to dose for the all pathways analysis, the air pathway analysis, the groundwater analysis, and the intruder analysis. Technical justification is provided for those radionuclides considered in detail in the analyses, and conversely, those not considered in the analyses.	Yes	In Section 3.2.1, the radionuclides relevant to the PA are identified by means of a reasonable screening analysis. All radionuclides produced at Hanford were considered. The inventory of potentially important radionuclides, Section 3.2.2, was developed from generally reasonable assumptions regarding radionuclide production in reactors and partitioning in chemical processing and in the waste tanks. However, the estimate of ^{99}Tc inventory appears to be considerably over stated (see detailed comments by H. Babad).	The Tc-99 inventory appears to be considerably overstated.	Same approach, see Section 3.2. H. Babad's comments were based on a contract signed after 1998 ILAW PA was submitted and on results from short-term experiments. These experiments have been continued, but results still are not definitive.

LFRG Comments on Performance Assessment for Hanford Immobilized Low-Activity Tank Waste, 7/27/99.				
Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.1.i. PA accounts for all relevant mechanisms for the release of radionuclides from the waste materials for environmental transport. The mechanisms analyzed are justified by references to relevant studies, available data, or supporting analyses in the PA.	No	The performance assessment presents a base case using a simplified model because the waste form had not been determined and only specifications for its short-term release rate are known (p. 3-8).The base case assumed the glass waste form released contaminants at a rate equivalent to the TWRS privatization RFP specification of 1.4×10^{-13} s^{-1} (p.3-39). It was assumed that this rate remained constant over time (however, the dimensions of the waste form decreased at a constant rate). Initially, the waste form was assumed to be a non-fractured monolith with dimensions of 1.2 x. 1.2 x. 1.8 m. No justification for the appropriateness of this assumption was provided. The preparers should have reviewed available data on glass leaching to put the assumed rate of glass corrosion into perspective. The contaminant release rates were independent of the infiltration rates used in the sensitivity study. This makes the infiltration sensitivity results less informative. The PA should discuss the vadose moisture relationship to contaminant release rate.	No evidence has been presented that the modeled performance has any relation to the actual performance which may reasonably be expected. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F). Need discussion of the relationship between vadose moisture and contaminant release rate.	Uses experimental data and calculations to simulate waste form performance, including the effect of varying recharge. See Section 4.6
3.1.1.j. PA provides a complete and clear description of the conceptual model of the environmental transport of the radionuclides from the waste materials to the points of compliance by air and water. The conceptual model is justified by referenced investigations, data, and supporting analyses that are representative of the site-specific conditions described.	Yes	With one exception, the conceptual model for the ground water pathway is complete, clearly described, and justified. See criterion 3.1.1.b for suggestions. The exception is the rate of contaminant release from the glass waste form. The specification in the privatization contract for waste immobilization is the short-term glass corrosion rate. No evidence is provided that the assumed rate can be achieved. The conceptual model for the air pathway is complete, clearly described, and justified.	Lack of evidence that assumed glass waste form performance can be achieved. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Uses experimental data and calculations to simulate waste form performance. See Sections 3.5.3.1 and 3.5.4.1

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Criteria	Criteria met?	Response	Issues	This Document
3.1.1.j.1. The conceptual model incorporates interpretations of available geochemical, geologic, meteorologic and hydrologic data, and the relevant mechanisms that have a significant effect on the transport of radionuclides at the disposal site.	Yes	Relevant mechanisms, supported by reasonable interpretations of available data are incorporated in the conceptual model for the air pathway. In general, the use of only two soil types in the vadose-zone model prevents evaluations of heterogeneities such as uncased, lost boreholes or preferential flow paths. The PA should more thoroughly present information on recharge rates and the relationship to contaminant concentrations and flux. The conceptual model for the base case does not incorporate mechanisms of glass corrosion. Rather, an assumed corrosion rate is used with no evidence provided to indicate whether the corrosion rate can actually be achieved.	Unsupported glass corrosion rate. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Sensitivity cases for vadose zone added to address heterogeneity - clastic dyke, information on recharge rate and relationship to contaminant concentrations and flux. See Section 3.5.5. Glass corrosion mechanisms included in base analysis case (See Section 3.5.3.1).
3.1.1.j.2. Assumptions incorporated into the conceptual model to account for transport mechanisms lacking sufficient data or supporting analyses are identified and justified as reasonable representations of site behavior over the time period considered in the analysis.	No	Performance of glass waste form is assumed with no evidence that the assumption is credible.	No assurance that assumed glass performance can actually be achieved. .Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Uses experimental data and calculations to simulate waste form performance. See Sections 3.4.4 and Appendix E

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Criteria	Criteria met?	Response	Issues	This Document
3.1.1.j.3. The conceptual model includes closure of the facility as justified based on referenced closure plans or reasonable assumptions of facility closure.	Yes	The conceptual model includes reasonable assumptions for facility closure. However, it is not clear exactly which closure design will be used (see pp. ES-xviii and 6.1). We recognize that conceptual designs were completed after completion of the PA.	No issues.	Closure based on issued Closure Plan (Burbank 2000) as approved by Boston 2000b. See Section 2.4.1
3.1.1.j.4. The conceptual model includes any credits to be taken in the analysis for the performance of engineered features. Credits for engineered features include a reasonable representation of the degradation of the engineered features that is justified by supporting investigations and data.	Yes	The analysis takes credit for the Hanford barrier to reduce infiltration. Increasing the infiltration rate to the natural rate after the first 1,000 years (the design life of the barrier) incorporated degradation of the barrier. A sensitivity case that used the natural recharge rate for the entire time provided information on the effect of a shorter life for the barrier than the design life. The analysis did not take credit for the waste packages. This is at least somewhat conservative. The analysis may be very conservative from the viewpoint that the waste containers will certainly be made of some sort of steel. Steel corrosion products are likely to be good at sorbing at least some radionuclides. This could have been investigated in a sensitivity case. The analysis takes credit for the concrete vaults, but only for 500 years. This is likely conservative. Sensitivity analyses were done with no credit for the vaults and with longer credit (i.e., 2,000 years) for the vaults.	No issues.	Near-field calculations involved full set of chemical reactions. Other work is similar to that in 1998 ILAW PA. See Sections 3.5.3.1 and 4.3.3.
3.1.1.j.5. The conceptual model includes natural processes that affect the transport of radionuclides (e.g., flooding, mass wasting, erosion, weathering) over the time period considered in the analysis, as justified based on referenced investigations and supporting analysis.	Yes	The conceptual model includes relevant natural processes and is consistent with other site PAs.	No issues.	Same approach. (See Sections 3.4 and 3.5)

LFRG Comments on Performance Assessment for Hanford Immobilized Low-Activity Tank Waste, 7/27/99.

Criteria	Criteria met?	Response	Issues	This Document
3.1.1.k. PA provides a clear description of the mathematical models used in the analyses. The mathematical models selected are justified and provide a reasonable representation of all of the elements of the conceptual model.	Yes	More clarification would add value to the analysis. For example, more information is required to describe the unit cell model used in some sensitivity analyses. What code is used (e.g., PORFLOW)? How many elements are used? What specific comparisons were made to the disposal facility model (e.g., moisture content, radionuclide transport, etc.) to ensure that these models are consistent?	Need additional information on unit cell model.	Unit cell no longer used. The combination of 1d and 2d calculations is discussed in Section 3.5.
3.1.1.k.1. The complexity of the mathematical models selected is commensurate with the available site data.	Yes	Mathematical models/computer code (AREST-CT for source release, PORFLOW for unsaturated zone, and VAM3D-CG for unsaturated flow and results) selections are commensurate with available data. Although, an observation regarding the difficulties and confusion due to many codes and models utilized in this PA, and to the non-integrated selection methodology to the codes/models in the CA (and other DOE disposal facilities) may lead to major inefficiencies.	No issues.	Updated codes were used (Section 3.5.2). Groundwater code/model is that used in CA.
3.1.1.k.2. Assumptions incorporated into the mathematical models are identified, justified, and consistent with the conceptual model	Yes	Assumptions are identified, justified, and consistent with the conceptual model.	No issues.	Same approach (See Section 3.5.2)
3.1.1.k.3. Mathematical models selected are documented and verified either in referenced publications or in the appendices of the performance assessment	Yes	Models are well documented.	No issues.	Same approach (See Section 3.5.2)

LFRG Comments on Performance Assessment for Hanford Immobilized Low-Activity Tank Waste, 7/27/99.				
Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.1.l. PA provides a complete description of the important exposure pathways and scenarios for the specific disposal facility that are used in the evaluation of the potential doses to the hypothetical, individual member of the public and inadvertent intruder consistent with site-specific environmental conditions and local and regional practices. The exposure pathways and scenarios selected for detailed analysis are justified as conservative representations of the long-term performance of the LLW disposal facility.	Yes	The PA provides a sufficiently complete description of pathways and scenarios (Section 3.3). With the exception of glass performance (see criterion 3.1.2.b), pathways and scenarios are reasonable and conservative representations of long-term performance.	Insufficient justification for assumed glass performance. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Uses experimental data and calculations to simulate waste form performance. See Sections 3.3.3. and 4.3.3.
3.1.1.l.1. Exposure pathways from the transport of contamination in groundwater that may be considered include potential exposures from the ingestion of contaminated groundwater, the use of contaminated groundwater for irrigation and livestock watering, and the biotic uptake and transport of contamination from groundwater and surface water. Potential exposure pathways from the transport of contamination in surface water include the ingestion of contaminated surface water and contaminated fish.	Yes	The exposure pathways from the transport of contamination in groundwater (Section 3.3.5) and include ingestion of contaminated groundwater, use of contaminated groundwater for irrigation of a small farm. Exposure comes from drinking contaminated water, ingesting contaminated food grown on the farm, ingesting and inhaling contaminated soil, and direct irradiation from the contaminated soil. Because the performance measure for surface water protection is a dose, assuming consumption of the surface water (Section 1.6.2.6), ingestion of contaminated fish was not considered.	No issues.	Same approach, see Section 3.3.5.

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3.1.1.1.2. If radiation dose is used as a measure of groundwater resource protection, the exposure scenarios consider the ingestion of water (at 2 liters per day or an alternative rate, if a justification is included) at the point of assessment, which represents the location of maximum exposure from a well constructed and developed using current practices typical for the local area	Yes	The groundwater protection scenario used a consumption rate of 2 L/day and assumed the well would be located to provide the maximum dose outside the 100-m buffer zone. Based on characterizations at numerous uranium mill tailings sites where inorganic contaminants were released decades ago and migrated into ground water, often in silty/sandy/gravely environments not unlike Hanford; vertical distribution of contaminant profile is often up to 20 meters. Based on these experiences, a 4.6-meter screen length seems reasonable and appropriate for the POC well 100-M down gradient of the disposal site (Section 3.4.7.2).	No issues.	Same approach. 4.6 m screen length justified based on local use (see Section 3.4.7)
3.1.1.1.3. Exposure scenarios from the transport of contamination in water for the all pathways analysis considers the use of groundwater and surface water consistent with local and regional practices. Exposure scenarios that may be considered include drinking water, crop irrigation and livestock watering, the ingestion of dairy products, livestock, fish, crops, and soil, the inhalation of resuspended particles, and external exposure.	Yes	The all pathways analysis considered groundwater uses consistent with local and regional practices. Exposure pathways (Section 3.3.4) include drinking contaminated water, ingesting contaminated food, ingesting and inhaling contaminated soil, and direct irradiation from the contaminated soil. The exposure pathways and scenarios are consistent with other Hanford performance assessments.	No issues.	Same approach (see Section 3.3.4)
3.1.1.1.4. Exposure pathways from the transport of contamination in the atmosphere that may be considered include potential exposure from immersion in air contaminated with volatile and nonvolatile radionuclides, deposition of volatile and nonvolatile radionuclides, and subsequent exposure from direct radiation, ingestion, and resuspension.	Yes	The analysis of potential exposure from the air pathway (Section 4.12) considered a limited set of radionuclides (^{3}H, ^{14}C, and ^{222}Rn) and a limited set of exposure pathways (inhalation and immersion). These assumptions are reasonable and are consistent with other Hanford PAs.	No issues.	Same approach (see Section 4.12)

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3.1.1.1.5. Exposure scenarios from the transport of contamination in air that may be considered include residential and gardening activities which include the direct inhalation of volatile and nonvolatile radionuclides, external exposure, ingestion of crops, soil, livestock, dairy products, and inhalation of resuspended particles.	Yes	Exposure scenarios for the transport of contamination in air are discussed in Section 4.12. Based on past performance assessments at Hanford, the analysis of airborne contamination is limited to three radionuclides (^{3}H, ^{14}C, and ^{222}Rn). The scenarios include inhalation and immersion in contaminated air.	No issues.	Same approach (see Section 4.12)
3.1.1.1.6. Exposure pathways from inadvertent intrusion into the waste disposal units identify the chronic and acute exposure pathways for each of the exposure scenarios considered. The exposure pathways include all relevant ingestion, external exposure, and inhalation pathways for each exposure scenario. [Direct ingestion of contaminated groundwater and exposures to radon should not be considered for inadvertent intrusion, because they are considered separately.]	Yes	The analysis of inadvertent intrusion (Section 5) identified the chronic and acute exposure pathways considered in the exposure scenarios. The chronic scenario was the homesteader scenario while for acute exposure, the well driller scenario was used. The exposure scenarios include all relevant pathways; direct ingestion of contaminated groundwater and exposures to radon are not considered.		Same approach (see Section 5)
3.1.1.1.7. Acute exposure scenarios for inadvertent intrusion considers direct intrusion into the disposal site and exhumation of accessible waste material. Relevant scenarios that may be considered include discovery, residential construction, and well drilling that incorporate external exposure, inhalation of resuspended particles, and ingestion of particles.	Yes	The acute scenario considered well drilling, direct intrusion into the disposal site and the exhumation of waste. The analysis properly rejected consideration of construction scenarios based on the depth of disposed waste. The drilling scenario analyzed included external exposure, inhalation of resuspended particles, and ingestion of particles.	No issues.	Same approach (see Section 5)

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3.1.1.1.8. Chronic exposure scenarios for inadvertent intrusion consider direct intrusion into the disposal site and exhumation of accessible waste material. Relevant scenarios that may be considered include residential use and post-construction, and post drilling agricultural use, that incorporate the ingestion of foodstuffs, ingestion of soil, external exposure, and inhalation of resuspended particles.	Yes	The chronic exposure scenario considered, the homesteader scenario (Section 5.2), includes direct intrusion into the disposal site and exhumation of waste. This scenario properly considered removal of waste by drilling rather than construction. The analysis included all relevant pathways and is consistent with other Hanford performance assessments.	No issues.	Same approach (see Section 5.3)
3.1.1.m. PA provides a coherent presentation of the relevant descriptive information concerning the site, the disposal facility, the waste characteristics that are reflected in the conceptual model, and the selection of the mathematical models used in the analysis. The descriptive information and the approach to modeling provide the necessary results to evaluate the exposure pathways and scenarios that are important to assess the performance of the disposal facility.	Yes	The description of the site, disposal facility, waste characteristics, and mathematical models are presented in a complete and coherent manner. The results are presented in a complete manner. However, no supporting information is provided to indicate that the assumed glass performance can actually be achieved.	No evidence that assumed glass performances can be expected to be achieved. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Same approach, except for glass. This PA uses experimental data and calculations to simulate waste form performance (see Sections 3.3.3, 3.4.4, and 3.5.3.1. for glass.)

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3.1.1.n. The calculated results presented in the PA are consistent with the site characteristics, the waste characteristics, and the conceptual model of the facility. The demonstration of consistency is supported by available site monitoring data and supporting field investigations.	Yes	The calculated results are presented in a thorough and complete manner. (See sections 3, 4, and 5. The results are generally consistent with site and waste characteristics. However, no evidence is provided that the long term performance of the glass waste form can actually be achieved.	Insufficient justification for assumed glass performance. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Same approach, except for glass. This PA uses experimental data and calculations to simulate waste form performance. See Sections 3.3.3, 3.4.4, and 3.5.3.1. for glass.
3.1.1.o. The models used for calculating the results presented in the PA are analyzed to identify the sensitive parameters in the analysis. The results of the sensitivity analysis are used to evaluate the uncertainty in the calculated results.	No	A deterministic base case was performed with a number of sensitivity analyses performed on a sub-system level (e.g., vadose-zone transport). However, some of the sensitivity analyses were performed with a different sub-system model (e.g., unit-cell model instead of the disposal facility model). The lack of consistency raises concern over the results and conclusions of the sensitivity analyses. No information was presented to relate the results of the unit cell model with the disposal facility model.	Lack of modeling consistency raises concern over the results and conclusions of the sensitivity analyses. In the factual accuracy review of the draft Review Team Report, Hanford disagreed with this comment.	No longer use unit cell model. Base case and sensitivity cases use same model. See Section 3.5.3.

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3.1.1.p. The results of the uncertainty analysis are interpreted as they relate to establishing reasonable assurance that the conclusions of the PA are correct.	Yes	The results of the uncertainty analysis are used in relation to reasonable assurance that conclusions are correct. However, no information is presented to indicate the assumed glass corrosion rate can actually be achieved.	Need assurance that assumed glass corrosion rates can actually be achieved. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Same approach, except for glass. This PA uses experimental data and calculations to simulate waste form performance. See Sections 3.3.3, 3.4.4, and 3.5.3.1.
3.1.1.q. The PA integrates the results of the analysis, the uncertainty analysis, the performance measures, waste acceptance criteria, operating procedures, and applicable laws, regulations, policies and agreements to formulate conclusions.	No	The PA integrates the results of the analysis, the uncertainty analysis, the performance measures, etc. to formulate conclusions. However, the conclusions of the PA regarding compliance with the groundwater protection performance objective is not supported because no evidence is provided that the assumed glass waste form performance can actually be achieved.	No support for assumed glass waste form performance. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Same approach, except for glass. This PA uses experimental data and calculations to simulate waste form performance. See Sections 3.3.3, 3.4.4, and 3.5.3.1.

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3.1.1.r. The PA includes an interpretation of the results that allows for a comparison to the performance measures used in the PA, and include any necessary limitations on facility design or operations that are required to meet the performance objectives.	Yes	The PA includes an interpretation of results compared to the Performance Objectives stated in Section 1.6. Section 6.3 discusses the requirements set by the results. Each performance objective discussion included the estimated impact compared to the base objective. Most discussions also included a list of key drivers or conditions that could affect the results.	No issues.	Same approach. See Section 7.2.
3.1.1.s. The PA discusses quality assurance measures applied to the preparation of the analysis and its documentation.	Yes	The description of quality assurance programs (Section 7) includes an overview description of the quality programs at various organizations that are involved in the ILAW project. In addition, the Hanford Environmental Dose Overview Panel reviewing approved dose calculations presented in the PA. Consequently, the discussion of quality programs is deemed complete.	No issues.	Same approach. See Appendix F
3.1.1.t. The PA includes an ALARA analysis, and if appropriate, the analytical methods for the ALARA assessment are described.	No	There is a minimal description of ALARA and similar activities in Section 4.11.	No issues.	Same approach. See Section 4.15
3.1.1.u. The PA included appendices or references to published documents and/or data that provide a basis for the discussions and analysis in the PA.	Yes	The appendices and references are complete to a fault. (See Section 9 and Appendices A-H).There is entirely too much reliance throughout the document on the references. For example, the general discussion on subsurface geology seems reasonable, however, details are vague. More traceable details should be present such as borehole logs and well completion drawings of the three new bore holes. Tables should be included that list pertinent boreholes and monitor wells with depth, toc, etc. Figure 2-10 poorly represents borehole locations. Consider using: a fold-out plate map, identify latitude and longitude coordinate system, fence diagram to illustrate geology in area of disposal.	Vagueness in the document as a result of over-reliance on reference material for important data that should be summarized in PA. In the factual accuracy review of the draft review team report, Hanford disagreed with this comment.	Same approach. However, we have included more details in main text.

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3.1.2. PA is Thorough and Technically Supported				
3.1.2.a. The PA presents an estimate of the radionuclide inventory of the radioactive waste disposed of and to be disposed of at the facility which is quantified and technically supported by records, data, studies and evaluations.	Yes	The waste being considered in this PA is from the Hanford high-level waste tanks. The waste was produced from chemical separation of fuel and target elements irradiated in the Hanford reactors. The inventory first considered all radionuclides produced at Hanford. Then, a simple and defensible screening methodology was employed to determine the most important radionuclides from the perspective of doses calculated in a PA. The resulting set of important radionuclides (p. 3-2) is consistent with previous Hanford PAs (i.e., those for the 200-E and 200-W burial grounds). The inventory of potentially important radionuclides was developed from generally reasonable assumptions regarding radionuclide production in reactors and partitioning in chemical processing and in the waste tanks. However, the estimate of ^{99}Tc inventory is likely to be considerably over stated (see detailed comments by H. Babad in Appendix).	No issues.	Same approach, see Section 3.2. H. Babad's comments were based on a contract signed by 1998 ILAW PA was submitted and on short-term experimental data. Although the experiments have been continued, the results are not definitive.
3.1.2.a.1. All of the radionuclides disposed and anticipated to be present in wastes to be disposed of are evaluated in the performance assessment. Any radionuclides screened from detailed analysis or having no inventory limit are identified, and the bases for these conclusions are supported and defensible	Yes	Same as Section 3.1.2.a response.	Same as Section 3.1.2.a issues.	See above

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3.1.2.a.2. Any estimates of the radionuclide inventory for past waste disposals are described and to the extent practical are based on past waste disposal records, a reasonable expectation of actual waste content based on a knowledge of the processes that generated the waste, calculations, sampling data, technical studies, and reasonable projections of waste to be disposed	Yes	Same as Section 3.1.2.a response..	Same as Section 3.1.2.a issues.	See above

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3.1.2.b. The physical and chemical characteristics of the waste disposed of in the past that affect the release and transport of radionuclides are identified. The physical and chemical characteristics of the waste form are quantified and supported by laboratory or field studies, or are based on referenced documentation.	No	The physical and chemical characteristics of the waste form were not quantified and do not seem to be supported by laboratory or field studies or referenced documentation. The base case simply assumed the glass waste form released contaminants at a rate equivalent to the TWRS privatization RFP specification of $1.4x10^{-13}$ s^{-1} (p.3-39). It was assumed that this rate remained constant over time (however, the dimensions of the waste form decreased at a constant rate). Initially, the waste form was assumed to be a non-fractured monolithic 1.2 x. 1.2 x. 1.8 m cube. No justification for the appropriateness of this assumption was provided. The preparers should have reviewed available data on glass leaching to put the assumed rate of glass corrosion into perspective. However, no attempt was made to relate the composition or properties of this glass with the glass that will be disposed in this disposal facility. The physical characteristics of the waste were taken into account in the release model, but it is not clear that they were conservatively simulated. The base case assumes that the glass is a monolith in the form of a cube (dimensions given above). A sensitivity case was analyzed where the waste form has the shape of a thin plate. The actual physical form of the waste will be highly dependent on conditions associated with pouring the glass into the container and handling of the filled containers. The glass may crack extensively. It is not clear that the cases analyzed bound the performance of an extensively cracked waste form. Sensitivity analyses consider a more mechanistic basis for performance of the waste form, assumed to be glass. Section 3.3.3 (page 3-20) contains a brief discussion of a scenario that would lead to release of radionuclides from a glass waste form. However, this scenario is not used in the analysis. Getter material (page 3-41) and waste conditioning layers (page 3-42) are mentioned but not used in the analysis.	The PA is only valid if the privatized contractor can demonstrate that the specified release rate can be met for both short and long time frames. There is no discussion of this aspect contained in the PA. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	This PA uses experimental data and calculations to simulate waste form performance. The ILAW program is committed to perform tests on the glass selected once the glass is selected. As noted in Section 7.5, it is highly likely that the glass selected will perform similarly to LAWABP1. Sensitivity case with high K_d for U and I provide insight into getter effect and water conditioning layer impact on waste form release is also discussed

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3.1.2.b. (continued) The physical and chemical characteristics of the waste disposed of in the past that affect the release and transport of radionuclides are identified. The physical and chemical characteristics of the waste form are quantified and supported by laboratory or field studies, or are based on referenced documentation.	No	Statements throughout the PA make clear that how release rate is controlled (i.e., the selection of waste forms and their performance) is the responsibility of the privatized contractor. Based on discussions with the PA analysts during the first Hanford site visit, the review team understood that the specific release rates in the waste form specification were derived from simple estimates of required disposal facility performance and not from capabilities of any specific waste forms. If this is true, then the base case analysis included in the PA does nothing more than confirm that the release rate selected for the contract specification was sufficiently small as to limit future doses to acceptable limits. This obviously circular argument leads to misleading statements, such as the one on page ES-ix. However, the authors recognize that more analysis must be done, as stated on page ES-xvii: The PA must show that these restrictions [e.g., release rates from waste forms] can be expected to be met. This task is attempted through the use of more mechanistic models contained in the sensitivity analyses. Some concerns related to the base case analysis include: The size assumed for the cubical waste form (page 3-42); The basis of equation 3.8 and its relationship to a cubic waste form (page 3-40; Basis for determining constant K4 (page 3-40); Basis for T=6.8E5 years (page 3-41); Concerns related to the sensitivity analysis include: Justification for spheres used as a waste form and their sizes (page C-3); Basis and justification for the release rate used in ARREST-CT (equation C.9); Basis and justification for the list of reactions used in the analysis (Table C.1); sensitivity analysis details that the fractional release rate will be met.	Same as Section 3.1.2.b. issues.	See above

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3.1.2.b. (continued) The physical and chemical characteristics of the waste disposed of in the past that affect the release and transport of radionuclides are identified. The physical and chemical characteristics of the waste form are quantified and supported by laboratory or field studies, or are based on referenced documentation.	No	In Appendix E (page E-42), a discussion is made claiming that use of computer models identified effects not seen in the laboratory (i.e., effects of ion exchange on pH). General concern that the model is giving correct information if it is not based on data derived from experimental sources. As a demonstration that the contract specification can be met, the PA is insufficient.	Same as Section 3.1.2.b. issues.	Note that the effect predicted has now been seen and measured (see Figure 3-8 in Section 3.4.4.3.)
3.1.2.c. Any inventory limits are developed from reasonable projections of waste to be disposed and analyses that consider the physical and chemical characteristics of the wastes if those characteristics affect the release and transport of the radionuclides.	No	Generally, the PA has identified significant parameters and has developed requirements for the facility and waste based on the analyses presented. Waste Acceptance Criteria (WAC) were developed from the analyses presented in the PA. However, as noted in criteria 3.1.1.i and 3.1.2.b, the assumed release rate of radionuclides from the glass waste form was that in the contract specification. No attempt was made to relate this assumed release rate to expected glass performance. In the analysis of compliance, the performance of glass LD6-5412 was assumed. There was no discussion of the performance of this glass in relation to that of glass expected to be produced.	No evidence is provided that the assumed glass waste form performance can be achieved. Thus, no assurance can be provided that performance objectives can reasonably be expected to not be exceeded. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	This PA uses experimental data and calculations to simulate waste form performance. See Sections 3.3.3, 3.4.4, and 3.5.3.1.

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3.1.2.d. The conceptual model is a reasonable interpretation of the existing geochemical geologic, meteorologic, hydrologic, and monitoring data for the site and disposal facility. The components of the conceptual model for the transport of radionuclides that are important to the conclusions relating to the long-term performance of the disposal facility are thoroughly analyzed. The assumptions incorporated into the conceptual model are consistent with the available data, related investigations, and theory related to the conceptual model. Any parameters included in the conceptual model are supported by data or related investigations relevant to the site and disposal facility.	Yes	Geochemical data is adequate (Table 3-8) and has adequate technical support. The sensitivity of this parameter has been adequately investigated for key radionuclides (e.g., page 4-50 *et seq.*). The geochemical data in the base case is adequately supported by data and investigations at the site, with appropriate references. The most mobile radionuclides (Sc, Tc) have been assigned a KD of O. Other radionuclides (U,I,C,S, and others) are assigned Kds less than 10 and at the bottom of the measured range for Hanford vadose zone soils. The sensitivity analysis examined both slight increases in the Kd for Se ad Tc and a decrease in the U and I Kd to zero. The sensitivity analysis for this parameter for these elements demonstrated that the dose from groundwater at 10,000 years is very dependent on this parameter for these elements. Finally, the long travel time of radionuclides in the groundwater results in significant radioactive decay of radionuclides with half lives less than 100 years. A table identifying calibration targets for recharge and head distributions for selected grids would be beneficial to the reader to better understand water balance, and observed vs. predicted fits. The water balance for the simulated post-Hanford conditions are provided in Chapter 3 (Section 3.5.3.3). Reference for the calibration fo the site-wdie model are provided in the same section.	No issues.	Same approach. Note that site-specific measurements indicate that I is mobile, while Se is not. See Section 3.4.3.3.

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3.1.2.e. The assumptions of the performance assessment related to the waste, site, and facility design and operations which are critical to the conclusions of the performance assessment are supported and the uncertainties associated with these assumptions are analyzed as part of the performance assessment. Credits for the performance of engineered features and site closure included in the conceptual model are based on data derived from field investigations, related investigations, or documented sources of information relevant to the site and disposal facility.	No	Assumptions related to the performance of the site and the facility are supported and uncertainties are analyzed. Credits for engineered features, including site closure, are derived from field investigations and other documented sources. However, as noted in criteria 3.1.1.i and 3.1.2.b, the assumed radionuclide release rate from the glass waste form is simply the contract specification for short-term release. There is no discussion of the relation of the assumed release rate and observed release rates of glass waste forms.	Noted in criteria 3.1.1.i and 3.1.2.b, the assumed radionuclide release rate from the glass waste form is simply the contract specification for short-term release. There is no discussion of the relation of the assumed release rate and observed release rates of glass waste forms. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	Same approach except for waste form. This PA uses experimental data and calculations to simulate waste form performance. See Sections 3.3.3, 3.4.4, and 3.5.3.1.
3.1.2.f. The conceptual model for the source term, groundwater flow, and radionuclide transport includes parameters for unsaturated and saturated flow, total and effective porosity, hydraulic conductivity, water retention, relative permeability relationships, volumetric water content, retardation, and diffusion that are based on data, related investigations, or documented references relevant to the site and disposal facility.	Yes	The vadose-zone parameters provided in Table 3-6 (p. 3-30) need further justification as to how they were obtained from Khaleel (1995). Were these parameters averaged over different soil types? See Section 3.1.2.d for geochemical discussions.		Same approach. The discussion in Section 3.4.3.2.2 has been expanded to address comment.

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3.1.2.g. The mathematical models used in the performance assessment for analyzing air and water transport of radionuclides are appropriate for the disposal facility and disposal site. The selected models provide a justified representation of the technically important mechanisms identified in the conceptual model, and provide calculated results that are a defensible basis for formulating conclusions.	Yes	Models used for the air pathways are appropriate and are consistent with the other Hanford performance assessments. PA should provide text on how radionuclide concentrations are transferred between the vadose and aquifer models. PA should consider that if the grid sizes are different, artificial dilution is occurring between the unsaturated- and saturated-zones.	Issues related to vadose zone modeling and interface between vadose and saturated zone matrix.	Same approach. Text added in Section 3.5.3.3.2.
3.1.2.g.1. The input data for the mathematical models are derived from field data from the site, laboratory data interpreted for field applications, or referenced literature sources which are applicable to the site. Assumptions which are used to formulate input data are justified and have a defensible technical basis.	Yes	The units on equations ES.3 & ES.4 (p. ES xv) are not consistent.		Units checked
3.1.2.g.2. Intermediate calculations are performed and results are presented that demonstrate, by comparison to site data or related investigations, the calculations of the mathematical models used in the performance assessment are representative of disposal site and facility behavior for important mechanisms represented in the mathematical models	Yes	The air pathway model is sufficiently simple that presentation of intermediate results is not appropriate. The unit cell model was used as a surrogate for the disposal facility model in many of the sensitivity analyses. However, there were no intermediate calculations to demonstrate that this simplified model correlates well with the disposal facility model. Therefore, sensitivity analyses may be providing results that are not consistent with the base case.	Correlation of unit cell model and disposal facility model is required to demonstrate that the sensitivity analyses are representative relative to the base case.	Same approach, except that unit cell is no longer used.

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3.1.2.g.3. Representations of groundwater well performance (e.g., construction, diameter, yield, depth of penetration, screen length) are reasonable reflections of regional practices and are justified.	No	Well completions might not be ideally suited to 3-d modeling. This potential data gap should be considered in future PA iterations.		Data covered in Section 3.4.7 based on tri-county survey.
3.1.2.g.4. The mathematical models are tested, by comparison to analytical calculations or other models, to demonstrate that the results are consistent with the conceptual model, physical and chemical processes represented in the models, and available site data. The models are evaluated for defensibility and are reasonable representations of the disposal site and facility performance by comparison to available site data, related technical investigations, or referenced documentation or literature.	Yes	The verification of mathematical models is discussed in Section 3.5. Models were verified versus other codes, analytical solutions, or site data, as appropriate. However, some questions regarding fundamental modeling assumptions remain. They are: Are vadose-zone moisture contents and water fluxes simulated at steady-state? If not, what are the initial conditions? If they are not steady, as the text seems to indicate, then how can the transport equations given in Eq. D.6 (p. D-3), which are based on steady-state flow, be used?	No issues.	Discused in Section 3.5.2.?
3.1.2.g.5. The initial conditions, the boundary conditions, and the changes of properties with time for the mathematical model are analytically correct (i.e., well posed), and derived from existing site data and information.	Yes	Boundary conditions of pumping model are taken from site-wide model, but these boundary conditions will change based on pumping rates used in the pumping model. Other boundary conditions are identified and reasonable.	No issues.	Same approach (See Section 3.5.3.3). Note pumping has small effect on aquifer.

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Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.2.h. The dose analysis considers the exposure pathways and transfer factors and calculates the maximum dose using acceptable methodologies and parameters.	Yes	The dose analysis considers exposure pathways and transfer factors using acceptable methodologies. The dose analysis of radionuclides uses transfer coefficients that are well supported by data and references. (See Appendix B and Section 5) The dose analysis specifies consumption rates, inhalation rates, and external exposure rates (i.e., occupancy) and conditions. The sources of these rates are identified and are justified. Section 3.4.7.1 discusses dose conversion factors, and Table B-7 shows the ratio of dose factor differences between DOE and EPA dose factors for ingestion. In reality, the EPA dose factors should be used, since they are the more recent compilation and were intended for government-wide use (See Section 6.3.2.2). As shown in the text (page 3-49 *et seq.*), this does not make a significant difference in the results. While some dose factors (e.g., Tc-99) are somewhat higher, this does not affect the results of the dose calculations in a significant way, as demonstrated by Table B-7. Consequently, the analysis is complete, albeit not with the use of the most appropriate dose factors. The maximum dose was projected for 10,000 years, rather than 1000 years. This period of analysis was selected because of Nuclear Regulatory Commission licensing requirements for incidental waste as produced during high level waste activities. Consequently, the decision to examine a 10,000 year period rather than a 1000 year period after closure is acceptable.	No issues.	Same approach, except that EPA dose factors are now used (See Section 3.4.7).
3.1.2.h. (continued) The dose analysis considers the exposure pathways and transfer factors and calculates the maximum dose using acceptable methodologies and parameters.	Yes	The methodologies and Parameters used are reasonable and are consistent with other Hanford performance assessments and are justified by the literature and site-specific investigations. It is noteworthy that, in the ILAW PA, the size of the garden has been reduced from 2,500 m^2 to 500 m^2. In the reviews of the earlier Hanford PAs, the larger garden size was controversial. The smaller size is reasonable and may be conservative.	Same as Section 3.1.2.h. issues.	See above.

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Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.2.h.1. The dose analysis for exposures to radionuclides identifies the transfer coefficients between media and justifies the parameters used in the analysis with supporting data or references to the literature	Yes	Same as Section 3.1.2.h response.	Same as Section 3.1.2.h. issues.	See above.
3.1.2.h.2. The dose analysis specifies the consumption of radioactively contaminated materials for the exposure pathways evaluated, the inhalation rates of contaminated materials, and the external exposure rates and conditions to radioactive materials. These parameters are justified using references to the literature or site-specific investigations	Yes	Same as Section 3.1.2.h response.	Same as Section 3.1.2.h. issues.	See above
3.1.2.h.3. The dose analysis is conducted using effective dose equivalents in accordance with ICRP-30 (1979) and uses dose conversion factors from recognized published sources.	Yes	Same as Section 3.1.2.h. response.	Same as Section 3.1.2.h. issues.	See above

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Criteria	Criteria met?	Response	Issues	This Document
3.1.2.h.4. The maximum dose projected for 1000 years after facility closure at the point of compliance is used in the analysis for evaluating disposal of LLW or establishing waste acceptance criteria for future disposals	Yes	The PA used 10,000 years instead of 1,000 years for the time of compliance. This was justified on the basis of being necessary to obtain NRC concurrence in the incidental waste determination that will be required by DOE 435.1.	No issues.	1,000 years is used for the time of compliance, but comparisons at 10,000 years also are given for NRC. See Section 1.6. Requirements are based on 10,000 year comparison.
3.1.2.i. The sensitivity and uncertainty analysis considers those parameters and mechanisms that are important to the conclusions relating to the long-term performance of the disposal facility, including radionuclide inventory, radionuclide characteristics, release rates, site and facility characteristics, groundwater flow parameters, site meteorology, and radionuclide transport parameters. Parametric and mechanistic variations analyzed in the uncertainty analysis that are important to the conclusions are justified as reasonable for the site and facility using data or related field investigations.	No	A deterministic base case was performed with a number of sensitivity analyses performed on a sub-system level (e.g., vadose-zone transport). However, the sensitivity analyses were occasionally performed with a different sub-system model (e.g., unit-cell model vs. disposal facility model). The lack of consistency raises concern over the results and conclusions of the sensitivity analyses. Additional sensitivity studies on hydrologic properties and strata types are needed. Specifically, the inclusion of heterogeneities in the vadose zone can lead to fast flow paths and should be considered in sensitivity analyses.	Lack of consistency between models used for the facility and sensitivity cases. Additional sensitivity analyses are needed to provide robustness to the analysis (e.g., vadose zone heterogeneities, poorer waste form performance).	Same sub-system models used throughout. See Section 3.5.3. More sensitivity studies run. See Section 3.5.5.

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Criteria	Criteria met?	Response	Issues	This Document
3.1.2.i.1. The parameters important to the components of the analysis are analyzed to identify the sensitive parameters, and the selection of sensitive parameters is quantitatively justified	No	Same as Section 3.1.2.i response.	Same as Section 3.1.2.i. issues.	See above
3.1.2.i.2. The sensitive parameters are analyzed for uncertainty in the results of the analysis to provide quantitative bounds for interpreting the results of the analysis.	No	Same as Section 3.1.2.i response.	Same as Section 3.1.2.i. issues.	See above
3.1.2.i.3. The results of the sensitivity analysis are determined using a prescribed methodology that is technically justified. The results of the analysis provide the necessary information to justify the assumptions and conclusions of the performance assessment.	No	Same as Section 3.1.2.i response.	Same as Section 3.1.2.i. issues.	See above
3.1.2.i.4. The maximum projected dose and time of occurrence is presented in the performance assessment to provide for understanding of the natural system being modeled and the behavior of the model.	No	Same as Section 3.1.2.i. response.	Same as Section 3.1.2.i. issues.	See above
3.1.2.j. The ALARA analysis provides a cost-benefit analysis that is an optimization of the collective or population dose based on the cost of dose reduction in the exposed population of $1,000 to $10,000 per person-rem averted. [ALARA analysis is not required if the projected individual or collective doses in the exposed population are trivial.]	No	Minimal ALARA discussion was presented. There is no explicit cost-benefit analysis relating to ALARA because conceptual facility designs were not available at the time of the report generation.	Minimal ALARA analysis.	Same approach, still do not have sufficient design information because of change to trench design.

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Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.2.k. The inadvertent intruder analysis considers the natural and man-made processes that impact the possible exposure to an intruder and calculates the dose using acceptable methodologies and parameters.	Yes	The inadvertent intruder analysis has considered reasonable natural and man-made processes that impact the possible exposure to an intruder. The dose resulting from the scenarios was calculated using acceptable methodologies and parameters. The analysis considers reduction in radionuclide concentration by mixing with soil in a garden. It is noteworthy that the ILAW PA used a more conservative value for the size of the garden than the value used in previous performance assessments (the value used in previous assessments was considered too large by some reviewers).	No issues.	Same approach. (See Section 5.)
3.1.2.k.1. The inadvertent intruder analysis specifies the reductions in concentrations of radioactive material from mixing with uncontaminated material or the transport of radionuclides from the disposed waste mass, and justifies the parameters used in the analysis with site data, supporting analysis or referenced information.	Yes	Same as Section 3.1.2.k response.	Same as Section 3.1.2.k issues.	See above
3.1.2.k.2. The inadvertent intruder analysis accounts for naturally occurring processes (e.g., erosion, precipitation, flooding) and the degradation of engineered barriers in the calculation of results	Yes	Same as Section 3.1.2.k response.	Same as Section 3.1.2.k issues.	See above

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Criteria	Criteria met?	Response	Issues	This Document
3.1.2.k.3. The inadvertent intruder analysis calculates the maximum dose from disposed materials during the period of 100–1000 years after site closure for waste acceptance criteria for wastes to be disposed of in the disposal facility using the recommendations of ICRP-30 (1979) and dose conversion factors from recognized published sources.	Yes	The inadvertent intruder analysis has assumed that the Hanford barrier used to cover the closed disposal facility remains intact for 1,000 years. The PA assumed that the concrete vaults would be degraded at 500 years. These assumptions are considered reasonable. The inadvertent intruder analysis presented the dose calculated over the 1,000 year period of compliance. The dose calculation used dose conversion factors from recognized published sources that are consistent with ICRP-30. The maximum dose is used to assess disposal facility performance and establish waste acceptance criteria.	No issues.	Same approach. See Section 5.
3.1.2.l. The results of the analyses for transport of radionuclides and the inadvertent intrusion into the disposal facility, and the sensitivity and uncertainty of the calculated results are comprehensive representations of the existing knowledge of the site and the disposal facility design and operations.	Yes	Based on the caveat of existing knowledge of the site and since building criteria into the facility design and waste form requirements, the results are a reasonable representation. Specific facility design criteria have not been fully defined such as: cover design, including the hydraulic diverter; getter usage; filler material, therefore leaving flexibility within the design to compensate for waste form deviations if necessary. This is not explicitly stated though implied and leads to concern since decisions have not been made.	Not all facility design factors have been selected.. If design criteria are different than the scenarios completed, PA changes will have to be made.	Same approach.

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Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.3. PA Conclusions Are Valid and Acceptable				
3.1.3.a. The performance assessment presents valid conclusions that demonstrate that the all-pathways analysis, air pathway analysis, groundwater resource protection analysis, and inadvertent intruder analysis meet the performance objectives of DOE Order 5820.2A.	No	The performance assessment conclusions are not valid due to the incomplete information regarding waste form performance. See criterion 3.1.2.b.	Waste form performance not demonstrated. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	This PA uses experimental data and calculations to simulate waste form performance. Comparisons show increased margin over the 1998 ILAW PA. See Sections 3.3.3, 3.4.4, and 3.5.3.1.
3.1.3.a.1. The all pathways performance objective of 25 mrem/year effective dose equivalent is met over the performance period of 1000 years for all radionuclides disposed of in the disposal facility.	No	Same as Section 3.1.3.a. response.	Same as Section 3.1.3.a. issues.	See above
3.1.3.a.2. The air pathways performance objective of 10 mrem/year effective dose equivalent is met over the performance period of 1000 years for all radionuclides disposed of in the disposal facility.	Yes	The air pathways performance objective is met over the 10,000-year period justified for this analysis.	No issues.	Same approach and results. See Sections 4.12 and 7.2.5.

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Criteria	Criteria met?	Response	Issues	This Document
3.1.3.a.3. The radon performance objective of an average flux of 20 pCi/m^2/s at the disposal surface or 0.5 pCi/L in air at the point of compliance is met over the performance period of 1000 years for all radionuclides disposed of in the disposal facility.	Yes	This measure is met by two entirely separate analytical approaches. First, there are insufficient radon precursors to generate significant radon. Second, the thick cover over the disposed waste reduces radon emissions by providing a thick cover that attenuates the diffusion of radon into the air. Consequently, the emissions of radon are trivial and the performance measure is met.	No issues	Same approach and results. See Sections 4.12 and 7.2.5.
3.1.3.a.4. The groundwater resource performance measures for all radionuclides to be disposed of in the disposal facility are met over the performance period of 1000 years at the prescribed point of compliance.	No	The conclusion that the groundwater protection performance objective is met is not valid because no evidence that the assumed glass waste form performance is likely to be realized is provided.	Glass waste form performance was not demonstrated. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	This PA uses experimental data and calculations to simulate waste form performance. Comparisons show increased margin over the 1998 ILAW PA. See Sections 3.3.3, 3.4.4, and 3.5.3.1. See Section 7.2 for comparison with performance objectives.

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Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.3.a.5. The inadvertent intruder performance objectives of 100 mrem/year effective dose equivalent for chronic exposure and 500 mrem effective dose equivalent for acute exposure are met within the disposal facility over the performance period of 1000 years.	Yes	The inadvertent intruder performance objectives of 100 mrem/year for chronic exposure and 500 mrem for acute exposure are met for the period from about 200 years to 1,000 years and beyond. The results for the homesteader scenario exceed the 100 mrem/year performance objective at 100 years following closure. However, the assumption that inadvertent intrusion will be prohibited for 500 years following closure by passive means (markers, etc.) is reasonable.	No issues.	Same approach and similar results. See Sections 5 and 7.2.2.
3.1.3.a.6. The condition that doses from the disposal of waste are ALARA has been demonstrated and incorporated into the design and operations of the disposal facility.	Yes	An adequate ALARA discussion has been presented based on the level of available detailed planning. No summary of ALARA considerations was presented; instead this considerations relied heavily on references.	Minimal ALARA analysis.	Same approach.

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Criteria	Criteria met?	Response	Issues	This Document
3.1.3.b. The performance assessment conclusions incorporate the findings of the calculated results for the all pathways analysis, air pathway analysis, groundwater resource protection analysis, inadvertent intruder analysis, and sensitivity and uncertainty analysis. The results are interpreted and integrated to formulate conclusions which are supported by the results and the uncertainties in the results.	No	The PA conclusions do not incorporate all results because the waste form performance is not demonstrated and therefore cannot be interpreted into facility design criteria. See discussion of 3.1.2.b. and 3.1.2.e.	Waste form performance not demonstrated. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	This PA uses experimental data and calculations to simulate waste form performance. Comparisons show increased margin over the 1998 ILAW PA. See Sections 3.3.3, 3.4.4, and 3.5.3.1. See Section 7.2 for comparison to performance objectives.

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Criteria	Criteria met?	Response	Issues	This Document
3.1.3.c. The conclusions of the performance assessment are applied to the facility design and operations. The resulting design constraints and limitations on operations can be reasonably accomplished at the disposal facility.	No	The results of the performance assessment have been interpreted to derive requirements on facility design, waste form and facility operations. These requirements were then examined in comparison with expected waste inventories, possible designs, etc. The conclusion is that the constraints can be readily met. However, the conclusion regarding waste form performance is not valid because no support for the assumed performance is provided.	No evidence provided that assumed waste form performance is achievable. Issue is being addressed by Hanford's additional research activities as noted in the supplemental information (see Appendix F).	This PA uses experimental data and calculations to simulate waste form performance. Comparisons show increased margin over the 1998 ILAW PA See Sections 3.3.3, 3.4.4, and 3.5.3.1. See Section 7.6 for requirements.

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Criteria	**Criteria met?**	**Response**	**Issues**	**This Document**
3.1.3.d. The conclusions of the performance assessment address and incorporate any constraints included in any Federal, state, and local statutes or regulations or agreements that impact the site design, facility design, or facility operations. The conclusions also address and incorporate any procedural or site documentation changes or constraints due to the results of the facility performance assessment. Reasonable assurance exists that these constraints and impacts are appropriately addressed in the performance assessment.	No	This criterion is met with the exception of the constraint of long-term glass performance. No evidence is provided that the assumed performance can be achieved.	Same as Section 3.1.3.c. issues.	This PA uses experimental data and calculations to simulate waste form performance. Comparisons show increased margin over the 1998 ILAW PA. See Sections 3.3.3, 3.4.4, and 3.5.3.1. Also see Section 7.5.

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Criteria	Criteria met?	Response	Issues	This Document
3.1.3.e. The analysis, results, and conclusions of the performance assessment provide both a reasonable representation of the disposal facility=s long-term performance and a reasonable expectation that the disposal facility will remain in compliance with DOE Order 5820.2A	No	This criterion is met with the exception of the constraint of long-term glass performance. No evidence is provided that the assumed performance can be achieved.	Same as Section 3.1.3.c. issues.	This PA uses experimental data and calculations to simulate waste form performance. Comparisons show increased margin over the 1998 ILAW PA. See Sections 3.3.3, 3.4.4, and 3.5.3.1.Also see Section 7.5.

Appendix B

Dosimetry Dose Factors

Table B-1. Radionuclides to be Considered and Their Half Lives.

Nuclide	Half life (years)	Short-lived progeny in equilibrium with parent
H-3	12.33	
Be-10	1.600E+06	
C-14	5,730	
Na-22	2.6019	
Si-32+D	329.56	P-32
Cl-36	300,992	
K-40	1.277E+09	
Ti-44+D	47.30	Sc-44
V-49	0.92539 (338 d)	
Mn-54	0.85454 (312.12 d)	
Fe-55	2.7299	
Co-60	5.2713	
Ni-59	74,999	
Ni-63	100.10	
Se-79	805,000	
Rb-87	4.800E+10	
Sr-90+D	28.149	Y-90
Zr-93	1.530E+06	
Nb-91	680	
Nb-93m	16.13	
Nb-94	20,300	
Mo-93	3,500	
Tc-99	211,097	
Ru-106+D	1.01736 (371.59 d)	Rh-106
Pd-107	6.50E+06	
Ag-108m+D	127.00	Ag-108 (0.087)

Table B-1. Radionuclides to be Considered and Their Half Lives.

Nuclide	Half life (years)	Short-lived progeny in equilibrium with parent
Cd-109	1.26653 (462.6 d)	
Cd-113m	14.10	
In-115	4.410E+14	
Sn-121m+D	54.998	Sn-121 (0.776)
Sn-126+D	246,000	Sb-126m, Sb-126 (0.14)
Sb-125	2.7299	
Te-125m	0.15880 (58 d)	
I-129	1.570E+07	
Cs-134	2.0619	
Cs-135	2.30E+06	
Cs-137+D	29.999	Ba-137m (0.9443)
Ba-133	10.520	
Pm-147	2.6233	
Sm-147	1.060E+11	
Sm-151	89.997	
Eu-150	35.798	
Eu-152	13.330	
Eu-154	8.5919	
Eu-155	4.680	
Gd-152	1.080E+14	
Ho-166m	1,200	
Re-187	5.000E+10	
Tl-204	3.7801	
Pb-205	1.520E+07	
Pb-210+D	22.300	Bi-210
Bi-207	32.198	
Po-209	102.0	

Table B-1. Radionuclides to be Considered and Their Half Lives.

Nuclide	Half life (years)	Short-lived progeny in equilibrium with parent
Po-210	0.37886 (138.38 d)	
Ra-226+D	1,600	Rn-222, Po-218, Pb-214, Bi-214, Po-214(0.9998)
Ra-228+D	5.7498	Ac-228
Ac-227+D	21.769	Th-227(0.9862), Fr-223(0.0138), Ra-223, Rn-219, Po-215, Pb-211, Bi-211, Tl-207(.99725), Po-211(.00275)
Th-228+D	1.9129	Ra-224, Rn-220, Po-216, Pb-212, Bi-212, Po-212(0.6406), Tl-208(0.3594)
Th-229+D	7,340	Ra-225, Ac-225, Fr-221, At-217, Bi213, Po-213(0.9784), Tl-209(0.0216)
Th-230	75,380	
Th-232	1.405E+10	
Pa-231	32,759	
U-232	69.799	
U-233	159,198	
U-234	245,694	
U-235+D	7.037E+08	Th-231
U-236	2.342E+07	
U-238+D	4.468E+09	Th-234, Pa-234m, Pa-234 (0.0013)
Np-237+D	2.140E+06	Pa-233
Pu-236	2.8999	
Pu-238	87.697	
Pu-239	24,110	
Pu-240	6,563	
Pu-241+D	14.350	U-237 (2.39E-05)
Pu-242	373,507	
Pu-244+D	8.000E+07	U-240 (0.9988), Np-240m, Np-240 (0.0012)
Am-241	432.70	
Am-242m+D	141.00	Am-242(0.9955), Np-238(0.0045)
Am-243+D	7,370	Np-239

Table B-1. Radionuclides to be Considered and Their Half Lives.

Nuclide	Half life (years)	Short-lived progeny in equilibrium with parent
Cm-242	0.44611 (162.94 d)	
Cm-243	28.499	
Cm-244	18.100	
Cm-245	8,500	
Cm-246	4,730	
Cm-247+D	1.600E+07	Pu-243
Cm-248	339,981	
Cm-250+D	11,300	Pu-246(0.25), Am-246(0.25), Bk-250(0.14)
Bk-247	1,394	
Cf-248	0.91294 (333.45 d)	
Cf-249	350.60	
Cf-250	13.080	
Cf-251	897.98	
Cf-252	2.6449	

Note: Parentheses show (1) half-lives that are normally given in days, and (2) branching ratios that differ from 1.00.

Table B-2. Decay Chains Actually Computed.

Zr-93	→	Nb-93m				
Mo-93	→	Nb-93m				
Sb-125	→ .230	Te-125m				
Pm-147	→	Sm-147				
Eu-152	→ .2792	Gd-152				
Pb-210	→	Po-210				
Po-209	→ .9974	Pb-205				
Ra-226	→	Pb-210	→	Po-210		
Ra-228	→	Th-228				
Th-230	→	Ra-226	→	Pb-210	→	Po-210
Th-232	→	Ra-228	→	Th-228		
Pa-231	→	Ac-227				
U-232	→	Th-228				
U-233	→	Th-229				
U-234	→	Th-230	→	Ra-226	→	Pb-210
U-235	→	Pa-231	→	Ac-227		
Pu-236	→	U-232	→	Th-228		
Pu-238	→	U-234				
Pu-241	→	Am-241	→	Np-237		
Pu-244	→	Pu-240				
Am-241	→	Np-237				
Am-242m	→ .827 → .173	Cm-242 Pu-242	→	Pu-238	→	U-234
Am-243	→	Pu-239				
Cm-242	→	Pu-238	→	U-234		

Table B-2. Decay Chains Actually Computed.

Cm-243	→	Pu-239				
	.0024 →	Am-243				
Cm-244	→	Pu-240				
Cm-245	→	Pu-241	→	Am-241	→	Np-237
Cm-247	→	Am-243				
Cm-250	→	Cf-250				
	.14 →	Cm-246				
Bk-247	.25 →	Am-243				
Cf-248	→	Cm-244	→	Pu-240		
Cf-249	→	Cm-245	→	Pu-241	→	Am-241
Cf-250	→	Cm-246				
Cf-251	→	Cm-247				
Cf-252	→	Cm-248				

Notes:

1) Decay times are assumed to be less than 1000 years so that the ingrowth of progeny with long half-lives can be ignored.

2) There is a slight increase in the Pu-238 and U-234 for the Am-242m decay chain that is not shown. This is a result of the low-probability alpha decay of Am-242m. The complete chain is, Am-242m(0.00455)--->Np-238--->Pu-238--->U-234.

Table B-3. No Water Infiltration Case: Intruder.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Ingest	Inhale
H-3	0.00	1.44E-03	1.42E-03	2.05E-05
Be-10	2.15E-01	1.79E-01	1.04E-01	7.55E-02
C-14	2.73E-03	4.69E-02	4.64E-02	4.46E-04
Na-22	2.39E+03	2.57E-01	2.56E-01	1.63E-03
Si-32+D	2.28E+00	4.64E-01	2.44E-01	2.20E-01
Cl-36	4.64E-01	7.20E-02	6.73E-02	4.67E-03
K-40	1.73E+02	4.16E-01	4.13E-01	2.65E-03
Ti-44+D	2.40E+03	6.43E-01	5.47E-01	9.64E-02
V-49	0.00	1.44E-03	1.36E-03	7.36E-05
Mn-54	9.08E+02	6.30E-02	6.16E-02	1.43E-03
Fe-55	0.00	1.41E-02	1.35E-02	5.74E-04
Co-60	2.75E+03	6.44E-01	5.98E-01	4.67E-02
Ni-59	0.00	4.95E-03	4.67E-03	2.82E-04
Ni-63	0.00	1.35E-02	1.28E-02	6.61E-04
Se-79	3.78E-03	1.95E-01	1.93E-01	2.10E-03
Rb-87	2.85E-02	1.10E-01	1.09E-01	6.89E-04
Sr-90+D	4.68E+00	3.45E+00	3.40E+00	5.29E-02
Zr-93	0.00	1.05E-01	3.69E-02	6.85E-02
Nb-91	2.30E+00	1.78E-02	1.16E-02	6.23E-03
Nb-93m	2.11E-02	1.78E-02	1.16E-02	6.23E-03
Nb-94	1.72E+03	2.47E-01	1.59E-01	8.83E-02
Mo-93	1.20E-01	3.61E-02	3.00E-02	6.06E-03
Tc-99	2.54E-02	3.42E-02	3.24E-02	1.78E-03
Ru-106+D	2.33E+02	7.11E-01	6.09E-01	1.02E-01
Pd-107	0.00	6.04E-03	3.31E-03	2.73E-03
Ag-108m+D	1.75E+03	2.30E-01	1.69E-01	6.04E-02
Cd-109	2.99E+00	3.15E-01	2.91E-01	2.43E-02
Cd-113m	1.30E-01	3.90E+00	3.58E+00	3.26E-01
In-115	8.04E-02	4.31E+00	3.51E+00	7.98E-01
Sn-121m+D	4.28E-01	5.25E-02	5.00E-02	2.54E-03
Sn-126+D	2.14E+03	4.90E-01	4.69E-01	2.15E-02

Table B-3. No Water Infiltration Case: Intruder.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Ingest	Inhale
Sb-125	4.48E+02	6.50E-02	6.24E-02	2.60E-03
Te-125m	3.07E+00	8.31E-02	8.16E-02	1.56E-03
I-129	2.63E+00	6.17E+00	6.13E+00	3.71E-02
Cs-134	1.70E+03	1.64E+00	1.63E+00	9.88E-03
Cs-135	7.76E-03	1.58E-01	1.57E-01	9.71E-04
Cs-137+D	6.12E+02	1.12E+00	1.11E+00	6.81E-03
Ba-133	3.74E+02	7.72E-02	7.56E-02	1.67E-03
Pm-147	1.01E-02	3.17E-02	2.33E-02	8.36E-03
Sm-147	0.00	2.00E+01	4.11E+00	1.59E+01
Sm-151	2.00E-04	1.50E-02	8.64E-03	6.40E-03
Eu-150	1.58E+03	1.99E-01	1.41E-01	5.72E-02
Eu-152	1.22E+03	1.91E-01	1.44E-01	4.71E-02
Eu-154	1.34E+03	2.73E-01	2.12E-01	6.10E-02
Eu-155	3.70E+01	4.28E-02	3.40E-02	8.83E-03
Gd-152	0.00	5.54E+01	3.58E+00	5.18E+01
Ho-166m	1.86E+03	3.44E-01	1.79E-01	1.65E-01
Re-187	0.00	2.23E-04	2.11E-04	1.16E-05
Tl-204	8.16E-01	7.52E-02	7.47E-02	5.14E-04
Pb-205	1.43E-03	3.71E-02	3.62E-02	8.36E-04
Pb-210+D	1.20E+00	1.22E+02	1.19E+02	2.94E+00
Bi-207	1.64E+03	1.26E-01	1.22E-01	4.27E-03
Po-209	3.58E+00	5.54E+01	5.29E+01	2.52E+00
Po-210	9.28E-03	4.42E+01	4.22E+01	2.01E+00
Ra-226+D	1.91E+03	3.14E+01	2.96E+01	1.83E+00
Ra-228+D	1.05E+03	3.31E+01	3.20E+01	1.08E+00
Ac-227+D	3.84E+02	1.76E+03	3.29E+02	1.43E+03
Th-228+D	1.68E+03	9.16E+01	1.80E+01	7.36E+01
Th-229+D	2.99E+02	5.50E+02	8.96E+01	4.61E+02
Th-230	2.42E-01	8.17E+01	1.22E+01	6.95E+01
Th-232	1.05E-01	4.11E+02	6.07E+01	3.50E+02
Pa-231	3.64E+01	5.09E+02	2.36E+02	2.73E+02

Table B-3. No Water Infiltration Case: Intruder.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Ingest	Inhale
U-232	1.81E-01	1.70E+02	2.91E+01	1.41E+02
U-233	2.74E-01	3.52E+01	6.42E+00	2.88E+01
U-234	8.12E-02	3.44E+01	6.29E+00	2.82E+01
U-235+D	1.50E+02	3.22E+01	5.93E+00	2.62E+01
U-236	4.32E-02	3.26E+01	5.98E+00	2.67E+01
U-238+D	2.41E+01	3.11E+01	5.96E+00	2.52E+01
Np-237+D	2.11E+02	2.14E+02	9.87E+01	1.15E+02
Pu-236	4.56E-02	5.69E+01	2.60E+01	3.09E+01
Pu-238	3.06E-02	1.55E+02	7.11E+01	8.36E+01
Pu-239	5.76E-02	1.70E+02	7.87E+01	9.15E+01
Pu-240	2.97E-02	1.70E+02	7.87E+01	9.15E+01
Pu-241+D	3.78E-03	3.28E+00	1.52E+00	1.76E+00
Pu-242	2.60E-02	1.62E+02	7.47E+01	8.77E+01
Pu-244+D	3.62E+02	1.60E+02	7.38E+01	8.60E+01
Am-241	8.88E+00	1.76E+02	8.09E+01	9.47E+01
Am-242m+D	1.33E+01	1.69E+02	7.82E+01	9.09E+01
Am-243+D	1.77E+02	1.75E+02	8.07E+01	9.39E+01
Cm-242	3.44E-02	6.25E+00	2.56E+00	3.69E+00
Cm-243	1.14E+02	1.21E+02	5.58E+01	6.55E+01
Cm-244	2.56E-02	9.78E+01	4.49E+01	5.29E+01
Cm-245	6.84E+01	1.80E+02	8.31E+01	9.71E+01
Cm-246	2.36E-02	1.78E+02	8.22E+01	9.62E+01
Cm-247+D	3.50E+02	1.64E+02	7.60E+01	8.83E+01
Cm-248	1.78E-02	6.54E+02	3.02E+02	3.52E+02
Cm-250+D	3.42E+02	3.73E+03	1.73E+03	2.01E+03

The "Internal" column is the sum of the "Inhale" and "Ingest" columns. External and internal doses are separated because the glass waste matrix will prevent a portion of the exhumed activity from contributing to the internal dose.

Table B-4. No Water Infiltration Case: Post-Intrusion Resident.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Garden	Ingest	Inhale
H-3	0.00	1.46E+02	1.46E+02	2.37E-02	5.58E-04
Be-10	2.41E+00	1.65E+01	8.21E+00	3.78E+00	4.50E+00
C-14	3.00E-02	7.20E+02	7.19E+02	1.65E+00	2.60E-02
Na-22	2.36E+04	3.05E+02	2.97E+02	8.17E+00	8.54E-02
Si-32+D	2.55E+01	4.39E+02	4.17E+02	8.86E+00	1.30E+01
Cl-36	4.78E+00	9.96E+04	9.96E+04	2.25E+00	2.56E-01
K-40	1.93E+03	5.16E+03	5.15E+03	1.49E+01	1.56E-01
Ti-44+D	2.68E+04	8.57E+01	6.02E+01	1.98E+01	5.71E+00
V-49	0.00	1.54E-01	1.16E-01	3.50E-02	3.09E-03
Mn-54	6.98E+03	2.64E+02	2.62E+02	1.54E+00	5.83E-02
Fe-55	0.00	1.28E+00	8.15E-01	4.34E-01	3.02E-02
Co-60	2.89E+04	1.86E+02	1.63E+02	2.04E+01	2.61E+00
Ni-59	0.00	4.68E+00	4.49E+00	1.70E-01	1.68E-02
Ni-63	0.00	1.28E+01	1.23E+01	4.66E-01	3.93E-02
Se-79	4.24E-02	1.24E+02	1.16E+02	7.05E+00	1.25E-01
Rb-87	3.20E-01	1.83E+02	1.79E+02	3.98E+00	4.10E-02
Sr-90+D	5.15E+01	2.00E+04	1.98E+04	1.21E+02	3.09E+00
Zr-93	5.03E-03	7.48E+00	2.03E+00	1.36E+00	4.09E+00
Nb-91	2.58E+01	2.65E+00	1.86E+00	4.23E-01	3.71E-01
Nb-93m	2.32E-01	2.61E+00	1.83E+00	4.14E-01	3.64E-01
Nb-94	1.93E+04	3.64E+01	2.54E+01	5.79E+00	5.27E+00
Mo-93	1.33E+00	4.47E+01	4.33E+01	1.09E+00	3.64E-01
Tc-99	1.69E-01	7.93E+02	7.92E+02	7.01E-01	6.28E-02
Ru-106+D	1.90E+03	2.29E+02	2.08E+02	1.61E+01	4.39E+00
Pd-107	0.00	3.52E+00	3.24E+00	1.20E-01	1.62E-01
Ag-108m+D	1.96E+04	3.02E+02	2.92E+02	6.15E+00	3.59E+00
Cd-109	2.58E+01	8.55E+02	8.45E+02	8.16E+00	1.11E+00
Cd-113m	1.42E+00	1.26E+04	1.24E+04	1.27E+02	1.89E+01
In-115	9.04E-01	3.61E+02	1.85E+02	1.28E+02	4.76E+01
Sn-121m+D	4.78E+00	1.12E+01	9.24E+00	1.81E+00	1.50E-01
Sn-126+D	2.41E+04	1.05E+02	8.71E+01	1.71E+01	1.28E+00
Sb-125	4.44E+03	3.57E+01	3.31E+01	2.48E+00	1.52E-01
Te-125m	7.80E+00	3.40E+00	2.71E+00	6.73E-01	2.10E-02
I-129	2.58E+01	6.70E+03	6.50E+03	1.95E+02	1.93E+00
Cs-134	1.62E+04	1.60E+03	1.55E+03	5.05E+01	5.00E-01
Cs-135	8.72E-02	1.75E+02	1.69E+02	5.73E+00	5.79E-02
Cs-137+D	6.80E+03	1.23E+03	1.19E+03	4.01E+01	4.01E-01
Ba-133	4.06E+03	3.43E+01	3.16E+01	2.66E+00	9.59E-02
Pm-147	1.00E-01	4.04E+00	2.85E+00	7.48E-01	4.38E-01
Sm-147	0.00	1.65E+03	5.52E+02	1.50E+02	9.51E+02
Sm-151	2.24E-03	1.85E+00	1.16E+00	3.14E-01	3.80E-01

Table B-4. No Water Infiltration Case: Post-Intrusion Resident.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Garden	Ingest	Inhale
Eu-150	1.76E+04	2.73E+01	1.89E+01	5.11E+00	3.38E+00
Eu-152	1.34E+04	2.69E+01	1.90E+01	5.12E+00	2.74E+00
Eu-154	1.44E+04	3.87E+01	2.77E+01	7.44E+00	3.50E+00
Eu-155	3.86E+02	5.97E+00	4.33E+00	1.15E+00	4.90E-01
Gd-152	0.00	3.70E+03	4.81E+02	1.30E+02	3.09E+03
Ho-166m	2.09E+04	4.05E+01	2.41E+01	6.54E+00	9.83E+00
Re-187	0.00	1.75E+00	1.74E+00	7.62E-03	6.85E-04
Tl-204	8.38E+00	6.22E+00	3.71E+00	2.49E+00	2.80E-02
Pb-205	1.61E-02	6.82E+00	5.45E+00	1.32E+00	4.99E-02
Pb-210+D	1.33E+01	2.47E+04	1.94E+04	5.11E+03	2.37E+02
Bi-207	1.83E+04	2.46E+01	2.00E+01	4.39E+00	2.52E-01
Po-209	4.01E+01	6.40E+03	4.33E+03	1.92E+03	1.50E+02
Po-210	4.79E-02	2.61E+03	1.85E+03	7.07E+02	5.49E+01
Ra-226+D	2.15E+04	4.79E+03	3.52E+03	1.15E+03	1.13E+02
Ra-228+D	1.40E+04	5.48E+03	3.54E+03	1.20E+03	7.40E+02
Ac-227+D	4.25E+03	1.13E+05	1.68E+04	1.18E+04	8.42E+04
Th-228+D	1.59E+04	4.97E+03	7.37E+02	5.52E+02	3.69E+03
Th-229+D	3.35E+03	3.49E+04	4.13E+03	3.27E+03	2.75E+04
Th-230	7.38E+00	5.16E+03	5.62E+02	4.45E+02	4.15E+03
Th-232	8.06E+02	2.61E+04	2.94E+03	2.29E+03	2.09E+04
Pa-231	4.78E+02	3.81E+04	1.17E+04	8.78E+03	1.76E+04
U-232	3.01E+03	1.38E+04	3.73E+03	1.15E+03	8.96E+03
U-233	3.21E+00	2.74E+03	8.05E+02	2.32E+02	1.70E+03
U-234	9.04E-01	2.68E+03	7.88E+02	2.27E+02	1.66E+03
U-235+D	1.66E+03	2.51E+03	7.44E+02	2.14E+02	1.55E+03
U-236	4.81E-01	2.54E+03	7.49E+02	2.16E+02	1.57E+03
U-238+D	2.61E+02	2.45E+03	7.46E+02	2.15E+02	1.49E+03
Np-237+D	2.30E+03	2.39E+04	1.38E+04	3.49E+03	6.66E+03
Pu-236	1.02E+01	3.60E+03	1.07E+03	8.49E+02	1.68E+03
Pu-238	3.43E-01	1.07E+04	3.11E+03	2.58E+03	4.97E+03
Pu-239	6.48E-01	1.18E+04	3.45E+03	2.87E+03	5.46E+03
Pu-240	3.34E-01	1.18E+04	3.45E+03	2.87E+03	5.46E+03
Pu-241+D	1.19E-01	2.31E+02	6.77E+01	5.66E+01	1.07E+02
Pu-242	2.92E-01	1.12E+04	3.28E+03	2.72E+03	5.23E+03
Pu-244+D	4.07E+03	1.11E+04	3.24E+03	2.69E+03	5.13E+03
Am-241	9.98E+01	1.23E+04	3.69E+03	2.95E+03	5.65E+03
Am-242m+D	1.47E+02	1.20E+04	3.60E+03	2.89E+03	5.51E+03
Am-243+D	1.99E+03	1.22E+04	3.68E+03	2.94E+03	5.60E+03
Cm-242	1.97E-01	2.52E+02	7.39E+01	5.39E+01	1.24E+02
Cm-243	1.27E+03	8.30E+03	2.43E+03	2.01E+03	3.86E+03
Cm-244	2.82E-01	6.65E+03	1.94E+03	1.61E+03	3.10E+03
Cm-245	7.69E+02	1.25E+04	3.65E+03	3.03E+03	5.80E+03

Table B-4. No Water Infiltration Case: Post-Intrusion Resident.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Garden	Ingest	Inhale
Cm-246	2.65E-01	1.23E+04	3.60E+03	3.00E+03	5.74E+03
Cm-247+D	3.93E+03	1.14E+04	3.33E+03	2.77E+03	5.27E+03
Cm-248	2.00E-01	4.53E+04	1.33E+04	1.10E+04	2.10E+04
Cm-250+D	3.85E+03	2.58E+05	7.57E+04	6.30E+04	1.20E+05
Bk-247	9.63E+02	1.58E+04	4.66E+03	3.81E+03	7.31E+03
Cf-248	2.04E-01	1.95E+03	1.24E+03	2.14E+02	4.99E+02
Cf-249	3.91E+03	3.38E+04	2.27E+04	3.84E+03	7.34E+03
Cf-250	2.63E-01	1.49E+04	1.00E+04	1.68E+03	3.25E+03
Cf-251	1.18E+03	3.46E+04	2.32E+04	3.93E+03	7.48E+03
Cf-252	3.53E-01	7.23E+03	4.70E+03	7.70E+02	1.76E+03

The "Internal" column is the sum of the "Garden", "Inhale", and "Ingest" columns. External and internal doses are separated because the glass waste matrix will prevent a portion of the exhumed activity from contributing to the internal dose.

Table B-5. Low Water Infiltration Case: All Pathways Farmer.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
H-3	4.58E-05	3.46E-05	7.04E-06	4.18E-06	0.00
Be-10	3.19E-03	2.52E-03	6.42E-04	3.03E-05	1.37E-06
C-14	4.67E-03	1.13E-03	3.54E-03	1.79E-07	1.71E-08
Na-22	5.21E-02	6.21E-03	3.22E-02	6.54E-07	1.37E-02
Si-32+D	7.42E-03	5.94E-03	1.38E-03	8.81E-05	1.44E-05
Cl-36	4.81E-02	1.64E-03	4.65E-02	1.87E-06	2.80E-06
K-40	2.83E-02	1.00E-02	1.72E-02	1.06E-06	1.10E-03
Ti-44+D	5.70E-02	1.33E-02	2.85E-02	3.87E-05	1.52E-02
V-49	4.26E-05	3.32E-05	9.42E-06	2.94E-08	0.00
Mn-54	6.19E-03	1.50E-03	4.11E-04	5.71E-07	4.28E-03
Fe-55	6.05E-04	3.28E-04	2.77E-04	2.30E-07	0.00
Co-60	4.57E-02	1.45E-02	1.46E-02	1.87E-05	1.66E-02
Ni-59	1.71E-04	1.13E-04	5.80E-05	1.13E-07	0.00
Ni-63	4.71E-04	3.12E-04	1.59E-04	2.65E-07	0.00
Se-79	1.15E-02	4.70E-03	6.80E-03	8.42E-07	2.40E-08
Rb-87	7.50E-03	2.66E-03	4.84E-03	2.76E-07	1.81E-07
Sr-90+D	1.19E-01	8.26E-02	3.59E-02	2.12E-05	2.93E-05
Zr-93	1.30E-03	8.96E-04	3.81E-04	2.75E-05	2.22E-09
Nb-91	3.06E-03	2.82E-04	2.76E-03	2.50E-06	1.46E-05
Nb-93m	3.01E-03	2.82E-04	2.73E-03	2.50E-06	1.32E-07
Nb-94	5.26E-02	3.86E-03	3.78E-02	3.54E-05	1.09E-02
Mo-93	1.19E-03	7.29E-04	4.62E-04	2.43E-06	7.55E-07
Tc-99	3.54E-03	7.88E-04	2.75E-03	7.09E-07	1.19E-07
Ru-106+D	1.98E-02	1.48E-02	3.78E-03	4.06E-05	1.15E-03
Pd-107	1.93E-04	8.05E-05	1.11E-04	1.10E-06	0.00
Ag-108m+D	2.51E-02	4.11E-03	9.89E-03	2.42E-05	1.11E-02
Cd-109	9.60E-03	7.07E-03	2.50E-03	9.72E-06	1.54E-05
Cd-113m	1.20E-01	8.69E-02	3.29E-02	1.31E-04	8.06E-07
In-115	1.27E-01	8.53E-02	4.17E-02	3.20E-04	5.11E-07
Sn-121m+D	4.54E-03	1.22E-03	3.32E-03	1.02E-06	2.71E-06
Sn-126+D	5.63E-02	1.14E-02	3.13E-02	8.64E-06	1.36E-02

Table B-5. Low Water Infiltration Case: All Pathways Farmer.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Sb-125	4.66E-03	1.52E-03	5.55E-04	1.04E-06	2.58E-03
Te-125m	2.77E-03	1.98E-03	7.86E-04	6.16E-07	5.68E-06
I-129	3.77E-01	1.49E-01	2.27E-01	1.49E-05	1.54E-05
Cs-134	1.10E-01	3.96E-02	6.07E-02	3.95E-06	9.49E-03
Cs-135	1.01E-02	3.82E-03	6.32E-03	3.89E-07	4.93E-08
Cs-137+D	7.53E-02	2.70E-02	4.44E-02	2.73E-06	3.85E-03
Ba-133	4.68E-03	1.84E-03	5.29E-04	6.68E-07	2.32E-03
Pm-147	7.70E-04	5.67E-04	2.00E-04	3.35E-06	5.81E-08
Sm-147	1.43E-01	9.99E-02	3.69E-02	6.39E-03	0.00
Sm-151	2.90E-04	2.10E-04	7.75E-05	2.57E-06	1.26E-09
Eu-150	1.47E-02	3.43E-03	1.26E-03	2.29E-05	9.99E-03
Eu-152	1.24E-02	3.50E-03	1.28E-03	1.89E-05	7.60E-03
Eu-154	1.53E-02	5.16E-03	1.88E-03	2.45E-05	8.23E-03
Eu-155	1.35E-03	8.26E-04	2.97E-04	3.54E-06	2.22E-04
Gd-152	1.36E-01	8.69E-02	2.82E-02	2.08E-02	0.00
Ho-166m	1.78E-02	4.36E-03	1.54E-03	6.61E-05	1.18E-02
Re-187	8.74E-06	5.14E-06	3.60E-06	4.65E-09	0.00
Tl-204	4.66E-03	1.81E-03	2.84E-03	2.06E-07	4.83E-06
Pb-205	1.12E-03	8.80E-04	2.39E-04	3.35E-07	9.10E-09
Pb-210+D	3.74E+00	2.90E+00	8.37E-01	1.19E-03	7.56E-06
Bi-207	1.42E-02	2.96E-03	8.82E-04	1.71E-06	1.04E-02
Po-209	1.83E+00	1.29E+00	5.41E-01	1.01E-03	2.27E-05
Po-210	1.41E+00	1.03E+00	3.81E-01	7.98E-04	3.17E-08
Ra-226+D	9.29E-01	7.18E-01	1.97E-01	7.36E-04	1.22E-02
Ra-228+D	9.97E-01	7.78E-01	2.11E-01	5.02E-04	7.68E-03
Ac-227+D	1.03E+01	7.99E+00	1.75E+00	5.75E-01	2.41E-03
Th-228+D	5.69E-01	4.38E-01	9.23E-02	2.95E-02	9.31E-03
Th-229+D	2.83E+00	2.18E+00	4.72E-01	1.85E-01	1.89E-03
Th-230	3.88E-01	2.96E-01	6.42E-02	2.79E-02	3.59E-06
Th-232	1.94E+00	1.47E+00	3.23E-01	1.40E-01	3.47E-04
Pa-231	7.08E+00	5.72E+00	1.25E+00	1.10E-01	2.62E-04
U-232	1.00E+00	7.07E-01	2.35E-01	5.65E-02	1.36E-03

Table B-5. Low Water Infiltration Case: All Pathways Farmer.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
U-233	2.19E-01	1.56E-01	5.13E-02	1.16E-02	1.80E-06
U-234	2.14E-01	1.53E-01	5.03E-02	1.13E-02	5.13E-07
U-235+D	2.03E-01	1.44E-01	4.74E-02	1.05E-02	9.45E-04
U-236	2.04E-01	1.45E-01	4.78E-02	1.07E-02	2.73E-07
U-238+D	2.03E-01	1.45E-01	4.76E-02	1.01E-02	1.48E-04
Np-237+D	2.97E+00	2.40E+00	5.25E-01	4.62E-02	1.32E-03
Pu-236	7.79E-01	6.32E-01	1.34E-01	1.24E-02	3.79E-06
Pu-238	2.14E+00	1.73E+00	3.74E-01	3.35E-02	1.94E-07
Pu-239	2.36E+00	1.91E+00	4.13E-01	3.67E-02	3.66E-07
Pu-240	2.36E+00	1.91E+00	4.13E-01	3.67E-02	1.89E-07
Pu-241+D	4.57E-02	3.70E-02	8.02E-03	7.06E-04	5.79E-08
Pu-242	2.24E+00	1.81E+00	3.92E-01	3.52E-02	1.65E-07
Pu-244+D	2.22E+00	1.79E+00	3.88E-01	3.45E-02	2.30E-03
Am-241	2.43E+00	1.97E+00	4.25E-01	3.80E-02	5.64E-05
Am-242m+D	2.35E+00	1.90E+00	4.13E-01	3.65E-02	8.33E-05
Am-243+D	2.42E+00	1.96E+00	4.24E-01	3.77E-02	1.12E-03
Cm-242	7.58E-02	6.21E-02	1.23E-02	1.47E-03	1.28E-07
Cm-243	1.68E+00	1.36E+00	2.95E-01	2.63E-02	7.20E-04
Cm-244	1.35E+00	1.09E+00	2.37E-01	2.12E-02	1.60E-07
Cm-245	2.50E+00	2.02E+00	4.41E-01	3.89E-02	4.35E-04
Cm-246	2.47E+00	2.00E+00	4.36E-01	3.86E-02	1.50E-07
Cm-247+D	2.29E+00	1.85E+00	4.03E-01	3.54E-02	2.22E-03
Cm-248	9.09E+00	7.34E+00	1.60E+00	1.41E-01	1.13E-07
Cm-250+D	5.19E+01	4.20E+01	9.16E+00	8.04E-01	2.17E-03
Bk-247	3.14E+00	2.54E+00	5.49E-01	4.91E-02	5.44E-04
Cf-248	2.43E-01	1.80E-01	5.87E-02	4.32E-03	1.24E-07
Cf-249	3.55E+00	2.56E+00	9.41E-01	4.94E-02	2.21E-03
Cf-250	1.59E+00	1.15E+00	4.19E-01	2.24E-02	1.50E-07
Cf-251	3.63E+00	2.62E+00	9.63E-01	5.03E-02	6.66E-04
Cf-252	8.01E-01	5.83E-01	2.04E-01	1.34E-02	2.05E-07

Table B-6. No Water Infiltration Case: Intruder.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Ingest	Inhale
H-3	0.00	1.44E-03	1.42E-03	2.05E-05
Be-10	2.15E-01	1.79E-01	1.04E-01	7.55E-02
C-14	2.73E-03	4.69E-02	4.64E-02	4.46E-04
Na-22	2.39E+03	2.57E-01	2.56E-01	1.63E-03
Si-32+D	2.28E+00	4.64E-01	2.44E-01	2.20E-01
Cl-36	4.64E-01	7.20E-02	6.73E-02	4.67E-03
K-40	1.73E+02	4.16E-01	4.13E-01	2.65E-03
Ti-44+D	2.40E+03	6.43E-01	5.47E-01	9.64E-02
V-49	0.00	1.44E-03	1.36E-03	7.36E-05
Mn-54	9.08E+02	6.30E-02	6.16E-02	1.43E-03
Fe-55	0.00	1.41E-02	1.35E-02	5.74E-04
Co-60	2.75E+03	6.44E-01	5.98E-01	4.67E-02
Ni-59	0.00	4.95E-03	4.67E-03	2.82E-04
Ni-63	0.00	1.35E-02	1.28E-02	6.61E-04
Se-79	3.78E-03	1.95E-01	1.93E-01	2.10E-03
Rb-87	2.85E-02	1.10E-01	1.09E-01	6.89E-04
Sr-90+D	4.68E+00	3.45E+00	3.40E+00	5.29E-02
Zr-93	0.00	1.05E-01	3.69E-02	6.85E-02
Nb-91	2.30E+00	1.78E-02	1.16E-02	6.23E-03
Nb-93m	2.11E-02	1.78E-02	1.16E-02	6.23E-03
Nb-94	1.72E+03	2.47E-01	1.59E-01	8.83E-02
Mo-93	1.20E-01	3.61E-02	3.00E-02	6.06E-03
Tc-99	2.54E-02	3.42E-02	3.24E-02	1.78E-03
Ru-106+D	2.33E+02	7.11E-01	6.09E-01	1.02E-01
Pd-107	0.00	6.04E-03	3.31E-03	2.73E-03
Ag-108m+D	1.75E+03	2.30E-01	1.69E-01	6.04E-02
Cd-109	2.99E+00	3.15E-01	2.91E-01	2.43E-02
Cd-113m	1.30E-01	3.90E+00	3.58E+00	3.26E-01
In-115	8.04E-02	4.31E+00	3.51E+00	7.98E-01
Sn-121m+D	4.28E-01	5.25E-02	5.00E-02	2.54E-03
Sn-126+D	2.14E+03	4.90E-01	4.69E-01	2.15E-02
Sb-125	4.48E+02	6.50E-02	6.24E-02	2.60E-03
Te-125m	3.07E+00	8.31E-02	8.16E-02	1.56E-03
I-129	2.63E+00	6.17E+00	6.13E+00	3.71E-02
Cs-134	1.70E+03	1.64E+00	1.63E+00	9.88E-03
Cs-135	7.76E-03	1.58E-01	1.57E-01	9.71E-04
Cs-137+D	6.12E+02	1.12E+00	1.11E+00	6.81E-03
Ba-133	3.74E+02	7.72E-02	7.56E-02	1.67E-03
Pm-147	1.01E-02	3.17E-02	2.33E-02	8.36E-03
Sm-147	0.00	2.00E+01	4.11E+00	1.59E+01
Sm-151	2.00E-04	1.50E-02	8.64E-03	6.40E-03

Table B-6. No Water Infiltration Case: Intruder.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Ingest	Inhale
Eu-150	1.58E+03	1.99E-01	1.41E-01	5.72E-02
Eu-152	1.22E+03	1.91E-01	1.44E-01	4.71E-02
Eu-154	1.34E+03	2.73E-01	2.12E-01	6.10E-02
Eu-155	3.70E+01	4.28E-02	3.40E-02	8.83E-03
Gd-152	0.00	5.54E+01	3.58E+00	5.18E+01
Ho-166m	1.86E+03	3.44E-01	1.79E-01	1.65E-01
Re-187	0.00	2.23E-04	2.11E-04	1.16E-05
Tl-204	8.16E-01	7.52E-02	7.47E-02	5.14E-04
Pb-205	1.43E-03	3.71E-02	3.62E-02	8.36E-04
Pb-210+D	1.20E+00	1.22E+02	1.19E+02	2.94E+00
Bi-207	1.64E+03	1.26E-01	1.22E-01	4.27E-03
Po-209	3.58E+00	5.54E+01	5.29E+01	2.52E+00
Po-210	9.28E-03	4.42E+01	4.22E+01	2.01E+00
Ra-226+D	1.91E+03	3.14E+01	2.96E+01	1.83E+00
Ra-228+D	1.05E+03	3.31E+01	3.20E+01	1.08E+00
Ac-227+D	3.84E+02	1.76E+03	3.29E+02	1.43E+03
Th-228+D	1.68E+03	9.16E+01	1.80E+01	7.36E+01
Th-229+D	2.99E+02	5.50E+02	8.96E+01	4.61E+02
Th-230	2.42E-01	8.17E+01	1.22E+01	6.95E+01
Th-232	1.05E-01	4.11E+02	6.07E+01	3.50E+02
Pa-231	3.64E+01	5.09E+02	2.36E+02	2.73E+02
U-232	1.81E-01	1.70E+02	2.91E+01	1.41E+02
U-233	2.74E-01	3.52E+01	6.42E+00	2.88E+01
U-234	8.12E-02	3.44E+01	6.29E+00	2.82E+01
U-235+D	1.50E+02	3.22E+01	5.93E+00	2.62E+01
U-236	4.32E-02	3.26E+01	5.98E+00	2.67E+01
U-238+D	2.41E+01	3.11E+01	5.96E+00	2.52E+01
Np-237+D	2.11E+02	2.14E+02	9.87E+01	1.15E+02
Pu-236	4.56E-02	5.69E+01	2.60E+01	3.09E+01
Pu-238	3.06E-02	1.55E+02	7.11E+01	8.36E+01
Pu-239	5.76E-02	1.70E+02	7.87E+01	9.15E+01
Pu-240	2.97E-02	1.70E+02	7.87E+01	9.15E+01
Pu-241+D	3.78E-03	3.28E+00	1.52E+00	1.76E+00
Pu-242	2.60E-02	1.62E+02	7.47E+01	8.77E+01
Pu-244+D	3.62E+02	1.60E+02	7.38E+01	8.60E+01
Am-241	8.88E+00	1.76E+02	8.09E+01	9.47E+01
Am-242m+D	1.33E+01	1.69E+02	7.82E+01	9.09E+01
Am-243+D	1.77E+02	1.75E+02	8.07E+01	9.39E+01
Cm-242	3.44E-02	6.25E+00	2.56E+00	3.69E+00
Cm-243	1.14E+02	1.21E+02	5.58E+01	6.55E+01
Cm-244	2.56E-02	9.78E+01	4.49E+01	5.29E+01
Cm-245	6.84E+01	1.80E+02	8.31E+01	9.71E+01

Table B-6. No Water Infiltration Case: Intruder.
Units are mrem per Ci exhumed.

Nuclide	External	Internal	Ingest	Inhale
Cm-246	2.36E-02	1.78E+02	8.22E+01	9.62E+01
Cm-247+D	3.50E+02	1.64E+02	7.60E+01	8.83E+01
Cm-248	1.78E-02	6.54E+02	3.02E+02	3.52E+02
Cm-250+D	3.42E+02	3.73E+03	1.73E+03	2.01E+03
Bk-247	8.56E+01	2.27E+02	1.04E+02	1.22E+02
Cf-248	2.53E-02	1.82E+01	7.42E+00	1.08E+01
Cf-249	3.48E+02	2.28E+02	1.05E+02	1.23E+02
Cf-250	2.40E-02	1.03E+02	4.73E+01	5.59E+01
Cf-251	1.05E+02	2.33E+02	1.08E+02	1.25E+02
Cf-252	3.56E-02	5.75E+01	2.40E+01	3.35E+01

The "Internal" column is the sum of the "Inhale" and "Ingest" columns. External and internal doses are separated because the glass waste matrix will prevent a portion of the exhumed activity from contributing to the internal dose.

Table B-7. Low Water Infiltration Case: HSRAM Industrial.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest	Inhale
H-3	1.62E-05	1.60E-05	2.02E-07
Be-10	1.17E-03	1.17E-03	7.43E-06
C-14	5.23E-04	5.23E-04	4.39E-08
Na-22	2.88E-03	2.88E-03	1.61E-07
Si-32+D	2.77E-03	2.75E-03	2.16E-05
Cl-36	7.58E-04	7.58E-04	4.60E-07
K-40	4.65E-03	4.65E-03	2.60E-07
Ti-44+D	6.16E-03	6.15E-03	9.49E-06
V-49	1.54E-05	1.54E-05	7.25E-09
Mn-54	6.93E-04	6.93E-04	1.41E-07
Fe-55	1.52E-04	1.52E-04	5.65E-08
Co-60	6.73E-03	6.73E-03	4.60E-06
Ni-59	5.25E-05	5.25E-05	2.77E-08
Ni-63	1.44E-04	1.44E-04	6.51E-08
Se-79	2.18E-03	2.18E-03	2.07E-07
Rb-87	1.23E-03	1.23E-03	6.78E-08
Sr-90+D	3.83E-02	3.83E-02	5.21E-06
Zr-93	4.22E-04	4.15E-04	6.74E-06
Nb-91	1.31E-04	1.31E-04	6.13E-07
Nb-93m	1.31E-04	1.31E-04	6.13E-07
Nb-94	1.79E-03	1.79E-03	8.69E-06
Mo-93	3.38E-04	3.38E-04	5.96E-07
Tc-99	3.65E-04	3.65E-04	1.75E-07
Ru-106+D	6.86E-03	6.85E-03	1.00E-05
Pd-107	3.75E-05	3.73E-05	2.69E-07
Ag-108m+D	1.91E-03	1.91E-03	5.94E-06
Cd-109	3.28E-03	3.28E-03	2.39E-06
Cd-113m	4.03E-02	4.03E-02	3.21E-05
In-115	3.96E-02	3.95E-02	7.85E-05
Sn-121m+D	5.63E-04	5.63E-04	2.50E-07
Sn-126+D	5.28E-03	5.28E-03	2.12E-06
Sb-125	7.03E-04	7.03E-04	2.56E-07
Te-125m	9.18E-04	9.18E-04	1.53E-07
I-129	6.90E-02	6.90E-02	3.65E-06
Cs-134	1.83E-02	1.83E-02	9.72E-07
Cs-135	1.77E-03	1.77E-03	9.56E-08
Cs-137+D	1.25E-02	1.25E-02	6.70E-07
Ba-133	8.50E-04	8.50E-04	1.64E-07
Pm-147	2.63E-04	2.63E-04	8.23E-07
Sm-147	4.78E-02	4.63E-02	1.57E-03
Sm-151	9.79E-05	9.73E-05	6.30E-07

Table B-7. Low Water Infiltration Case: HSRAM Industrial.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest	Inhale
Eu-150	1.60E-03	1.59E-03	5.63E-06
Eu-152	1.62E-03	1.62E-03	4.64E-06
Eu-154	2.39E-03	2.39E-03	6.01E-06
Eu-155	3.83E-04	3.83E-04	8.69E-07
Gd-152	4.54E-02	4.03E-02	5.10E-03
Ho-166m	2.03E-03	2.02E-03	1.62E-05
Re-187	2.38E-06	2.38E-06	1.14E-09
Tl-204	8.40E-04	8.40E-04	5.06E-08
Pb-205	4.08E-04	4.08E-04	8.23E-08
Pb-210+D	1.34E+00	1.34E+00	2.90E-04
Bi-207	1.37E-03	1.37E-03	4.20E-07
Po-209	5.95E-01	5.95E-01	2.48E-04
Po-210	4.75E-01	4.75E-01	1.97E-04
Ra-226+D	3.33E-01	3.33E-01	1.81E-04
Ra-228+D	3.60E-01	3.60E-01	1.07E-04
Ac-227+D	3.84E+00	3.70E+00	1.41E-01
Th-228+D	2.10E-01	2.03E-01	7.25E-03
Th-229+D	1.05E+00	1.01E+00	4.54E-02
Th-230	1.44E-01	1.37E-01	6.85E-03
Th-232	7.17E-01	6.83E-01	3.44E-02
Pa-231	2.68E+00	2.65E+00	2.69E-02
U-232	3.41E-01	3.28E-01	1.38E-02
U-233	7.51E-02	7.23E-02	2.84E-03
U-234	7.35E-02	7.08E-02	2.77E-03
U-235+D	6.93E-02	6.68E-02	2.58E-03
U-236	6.99E-02	6.73E-02	2.63E-03
U-238+D	6.95E-02	6.70E-02	2.48E-03
Np-237+D	1.12E+00	1.11E+00	1.13E-02
Pu-236	2.96E-01	2.93E-01	3.05E-03
Pu-238	8.08E-01	8.00E-01	8.23E-03
Pu-239	8.94E-01	8.85E-01	9.01E-03
Pu-240	8.94E-01	8.85E-01	9.01E-03
Pu-241+D	1.73E-02	1.71E-02	1.73E-04
Pu-242	8.49E-01	8.40E-01	8.63E-03
Pu-244+D	8.38E-01	8.30E-01	8.46E-03
Am-241	9.19E-01	9.10E-01	9.32E-03
Am-242m+D	8.89E-01	8.80E-01	8.95E-03
Am-243+D	9.17E-01	9.08E-01	9.24E-03
Cm-242	2.91E-02	2.88E-02	3.63E-04
Cm-243	6.34E-01	6.28E-01	6.45E-03
Cm-244	5.10E-01	5.05E-01	5.21E-03
Cm-245	9.45E-01	9.35E-01	9.56E-03

Table B-7. Low Water Infiltration Case: HSRAM Industrial.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest	Inhale
Cm-246	9.34E-01	9.25E-01	9.47E-03
Cm-247+D	8.64E-01	8.55E-01	8.69E-03
Cm-248	3.43E+00	3.40E+00	3.47E-02
Cm-250+D	1.96E+01	1.94E+01	1.97E-01
Bk-247	1.19E+00	1.18E+00	1.21E-02
Cf-248	8.46E-02	8.35E-02	1.06E-03
Cf-249	1.20E+00	1.19E+00	1.21E-02
Cf-250	5.38E-01	5.33E-01	5.50E-03
Cf-251	1.22E+00	1.21E+00	1.23E-02
Cf-252	2.73E-01	2.70E-01	3.30E-03

The "Total" column is the sum of the "Ingest" and "Inhale" columns.

Table B-8. Low Water Infiltration Case: HSRAM Residential.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
H-3	4.92E-05	4.67E-05	2.13E-06	2.98E-07	0.00
Be-10	3.89E-03	3.40E-03	4.79E-04	1.13E-05	2.32E-06
C-14	1.84E-03	1.53E-03	3.14E-04	6.69E-08	2.91E-08
Na-22	3.29E-02	8.40E-03	1.20E-03	2.44E-07	2.33E-02
Si-32+D	9.27E-03	8.03E-03	1.18E-03	3.30E-05	2.46E-05
Cl-36	1.64E-02	2.21E-03	1.42E-02	7.00E-07	4.77E-06
K-40	1.81E-02	1.36E-02	2.62E-03	3.97E-07	1.86E-03
Ti-44+D	4.63E-02	1.80E-02	2.53E-03	1.45E-05	2.58E-02
V-49	5.08E-05	4.48E-05	5.96E-06	1.10E-08	0.00
Mn-54	9.61E-03	2.02E-03	3.01E-04	2.13E-07	7.29E-03
Fe-55	5.04E-04	4.43E-04	6.11E-05	8.58E-08	0.00
Co-60	5.06E-02	1.96E-02	2.75E-03	7.00E-06	2.82E-02
Ni-59	1.75E-04	1.53E-04	2.22E-05	4.22E-08	0.00
Ni-63	4.82E-04	4.21E-04	6.08E-05	9.92E-08	0.00
Se-79	7.26E-03	6.35E-03	9.08E-04	3.15E-07	4.08E-08
Rb-87	4.12E-03	3.59E-03	5.29E-04	1.03E-07	3.08E-07
Sr-90+D	1.30E-01	1.12E-01	1.83E-02	7.93E-06	4.98E-05
Zr-93	1.39E-03	1.21E-03	1.71E-04	1.03E-05	3.77E-09
Nb-91	4.61E-04	3.81E-04	5.38E-05	9.34E-07	2.48E-05
Nb-93m	4.36E-04	3.81E-04	5.36E-05	9.34E-07	2.24E-07
Nb-94	2.45E-02	5.21E-03	7.36E-04	1.32E-05	1.85E-02
Mo-93	1.13E-03	9.86E-04	1.45E-04	9.09E-07	1.28E-06
Tc-99	1.31E-03	1.07E-03	2.44E-04	2.64E-07	2.02E-07
Ru-106+D	2.47E-02	2.00E-02	2.69E-03	1.52E-05	1.96E-03
Pd-107	1.25E-04	1.09E-04	1.57E-05	4.10E-07	0.00
Ag-108m+D	2.52E-02	5.56E-03	8.19E-04	9.06E-06	1.88E-02
Cd-109	1.10E-02	9.56E-03	1.40E-03	3.63E-06	2.63E-05
Cd-113m	1.36E-01	1.18E-01	1.81E-02	4.89E-05	1.37E-06
In-115	1.32E-01	1.15E-01	1.62E-02	1.20E-04	8.69E-07
Sn-121m+D	1.88E-03	1.64E-03	2.32E-04	3.81E-07	4.60E-06
Sn-126+D	4.07E-02	1.54E-02	2.18E-03	3.23E-06	2.32E-02
Sb-125	6.76E-03	2.05E-03	3.08E-04	3.90E-07	4.40E-03
Te-125m	2.97E-03	2.68E-03	2.84E-04	2.28E-07	9.67E-06
I-129	2.31E-01	2.01E-01	2.92E-02	5.55E-06	2.62E-05
Cs-134	7.72E-02	5.35E-02	7.59E-03	1.48E-06	1.61E-02
Cs-135	5.92E-03	5.16E-03	7.54E-04	1.46E-07	8.39E-08
Cs-137+D	4.84E-02	3.65E-02	5.32E-03	1.02E-06	6.56E-03
Ba-133	6.77E-03	2.48E-03	3.51E-04	2.50E-07	3.94E-03
Pm-147	8.74E-04	7.67E-04	1.06E-04	1.25E-06	9.89E-08
Sm-147	1.56E-01	1.35E-01	1.90E-02	2.39E-03	0.00
Sm-151	3.25E-04	2.84E-04	4.00E-05	9.60E-07	2.15E-09

Table B-8. Low Water Infiltration Case: HSRAM Residential.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Eu-150	2.23E-02	4.64E-03	6.54E-04	8.57E-06	1.70E-02
Eu-152	1.83E-02	4.73E-03	6.64E-04	7.07E-06	1.29E-02
Eu-154	2.20E-02	6.97E-03	9.77E-04	9.14E-06	1.40E-02
Eu-155	1.65E-03	1.12E-03	1.56E-04	1.32E-06	3.77E-04
Gd-152	1.42E-01	1.18E-01	1.66E-02	7.78E-03	0.00
Ho-166m	2.68E-02	5.89E-03	8.31E-04	2.47E-05	2.01E-02
Re-187	8.16E-06	6.94E-06	1.21E-06	1.74E-09	0.00
Tl-204	2.80E-03	2.45E-03	3.40E-04	7.70E-08	8.22E-06
Pb-205	1.36E-03	1.19E-03	1.68E-04	1.25E-07	1.55E-08
Pb-210+D	4.50E+00	3.92E+00	5.76E-01	4.46E-04	1.29E-05
Bi-207	2.22E-02	4.00E-03	5.64E-04	6.40E-07	1.76E-02
Po-209	1.98E+00	1.74E+00	2.44E-01	3.78E-04	3.86E-05
Po-210	1.56E+00	1.39E+00	1.71E-01	2.96E-04	5.39E-08
Ra-226+D	1.13E+00	9.71E-01	1.38E-01	2.75E-04	2.07E-02
Ra-228+D	1.21E+00	1.05E+00	1.49E-01	2.05E-04	1.31E-02
Ac-227+D	1.25E+01	1.08E+01	1.52E+00	2.15E-01	4.10E-03
Th-228+D	7.00E-01	5.92E-01	8.10E-02	1.10E-02	1.58E-02
Th-229+D	3.43E+00	2.94E+00	4.14E-01	6.91E-02	3.22E-03
Th-230	4.67E-01	4.00E-01	5.63E-02	1.04E-02	6.10E-06
Th-232	2.33E+00	1.99E+00	2.82E-01	5.25E-02	5.91E-04
Pa-231	8.87E+00	7.74E+00	1.09E+00	4.10E-02	4.45E-04
U-232	1.12E+00	9.56E-01	1.37E-01	2.11E-02	2.31E-03
U-233	2.45E-01	2.11E-01	2.97E-02	4.32E-03	3.07E-06
U-234	2.40E-01	2.07E-01	2.91E-02	4.22E-03	8.72E-07
U-235+D	2.28E-01	1.95E-01	2.75E-02	3.94E-03	1.61E-03
U-236	2.28E-01	1.96E-01	2.77E-02	4.00E-03	4.64E-07
U-238+D	2.27E-01	1.96E-01	2.76E-02	3.78E-03	2.52E-04
Np-237+D	3.72E+00	3.24E+00	4.57E-01	1.73E-02	2.24E-03
Pu-236	9.77E-01	8.54E-01	1.18E-01	4.63E-03	6.45E-06
Pu-238	2.68E+00	2.34E+00	3.28E-01	1.25E-02	3.30E-07
Pu-239	2.96E+00	2.58E+00	3.63E-01	1.37E-02	6.23E-07
Pu-240	2.96E+00	2.58E+00	3.63E-01	1.37E-02	3.21E-07
Pu-241+D	5.73E-02	5.00E-02	7.05E-03	2.64E-04	9.84E-08
Pu-242	2.81E+00	2.45E+00	3.45E-01	1.32E-02	2.81E-07
Pu-244+D	2.78E+00	2.42E+00	3.41E-01	1.29E-02	3.91E-03
Am-241	3.05E+00	2.66E+00	3.74E-01	1.42E-02	9.60E-05
Am-242m+D	2.95E+00	2.57E+00	3.62E-01	1.36E-02	1.42E-04
Am-243+D	3.04E+00	2.65E+00	3.73E-01	1.41E-02	1.91E-03
Cm-242	9.52E-02	8.40E-02	1.07E-02	5.47E-04	2.18E-07
Cm-243	2.10E+00	1.83E+00	2.57E-01	9.82E-03	1.23E-03
Cm-244	1.69E+00	1.47E+00	2.07E-01	7.93E-03	2.72E-07
Cm-245	3.13E+00	2.73E+00	3.84E-01	1.46E-02	7.40E-04

Table B-8. Low Water Infiltration Case: HSRAM Residential.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Cm-246	3.10E+00	2.70E+00	3.80E-01	1.44E-02	2.55E-07
Cm-247+D	2.86E+00	2.50E+00	3.51E-01	1.32E-02	3.78E-03
Cm-248	1.14E+01	9.93E+00	1.40E+00	5.28E-02	1.92E-07
Cm-250+D	6.50E+01	5.67E+01	7.98E+00	3.01E-01	3.70E-03
Bk-247	3.93E+00	3.43E+00	4.83E-01	1.84E-02	9.25E-04
Cf-248	2.79E-01	2.44E-01	3.31E-02	1.61E-03	2.10E-07
Cf-249	3.97E+00	3.46E+00	4.89E-01	1.85E-02	3.76E-03
Cf-250	1.78E+00	1.55E+00	2.19E-01	8.38E-03	2.55E-07
Cf-251	4.06E+00	3.54E+00	5.00E-01	1.88E-02	1.13E-03
Cf-252	9.03E-01	7.88E-01	1.09E-01	5.01E-03	3.49E-07

Table B-9. Low Water Infiltration Case: HSRAM Agricultural.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
H-3	5.73E-05	4.67E-05	1.03E-05	2.98E-07	0.00
Be-10	3.99E-03	3.40E-03	5.79E-04	1.13E-05	2.32E-06
C-14	6.44E-03	1.53E-03	4.92E-03	6.69E-08	2.91E-08
Na-22	8.82E-02	8.40E-03	5.65E-02	2.44E-07	2.33E-02
Si-32+D	9.30E-03	8.03E-03	1.22E-03	3.30E-05	2.46E-05
Cl-36	5.81E-02	2.21E-03	5.59E-02	7.00E-07	4.77E-06
K-40	4.09E-02	1.36E-02	2.55E-02	3.97E-07	1.86E-03
Ti-44+D	8.79E-02	1.80E-02	4.41E-02	1.45E-05	2.58E-02
V-49	5.31E-05	4.48E-05	8.25E-06	1.10E-08	0.00
Mn-54	9.73E-03	2.02E-03	4.18E-04	2.13E-07	7.29E-03
Fe-55	7.20E-04	4.43E-04	2.77E-04	8.58E-08	0.00
Co-60	6.57E-02	1.96E-02	1.79E-02	7.00E-06	2.82E-02
Ni-59	2.22E-04	1.53E-04	6.90E-05	4.22E-08	0.00
Ni-63	6.11E-04	4.21E-04	1.89E-04	9.92E-08	0.00
Se-79	1.44E-02	6.35E-03	8.10E-03	3.15E-07	4.08E-08
Rb-87	1.14E-02	3.59E-03	7.78E-03	1.03E-07	3.08E-07
Sr-90+D	1.60E-01	1.12E-01	4.81E-02	7.93E-06	4.98E-05
Zr-93	1.60E-03	1.21E-03	3.74E-04	1.03E-05	3.77E-09
Nb-91	3.91E-03	3.81E-04	3.50E-03	9.34E-07	2.48E-05
Nb-93m	3.84E-03	3.81E-04	3.46E-03	9.34E-07	2.24E-07
Nb-94	7.17E-02	5.21E-03	4.79E-02	1.32E-05	1.85E-02
Mo-93	1.56E-03	9.86E-04	5.72E-04	9.09E-07	1.28E-06
Tc-99	5.83E-03	1.07E-03	4.77E-03	2.64E-07	2.02E-07
Ru-106+D	2.54E-02	2.00E-02	3.44E-03	1.52E-05	1.95E-03
Pd-107	3.13E-04	1.09E-04	2.04E-04	4.10E-07	0.00
Ag-108m+D	4.37E-02	5.56E-03	1.93E-02	9.06E-06	1.88E-02
Cd-109	1.27E-02	9.56E-03	3.06E-03	3.63E-06	2.63E-05
Cd-113m	1.58E-01	1.18E-01	4.06E-02	4.89E-05	1.37E-06
In-115	1.55E-01	1.15E-01	4.00E-02	1.20E-04	8.69E-07
Sn-121m+D	5.19E-03	1.64E-03	3.54E-03	3.81E-07	4.60E-06
Sn-126+D	7.19E-02	1.54E-02	3.33E-02	3.23E-06	2.32E-02
Sb-125	6.99E-03	2.05E-03	5.35E-04	3.90E-07	4.40E-03
Te-125m	3.28E-03	2.68E-03	5.96E-04	2.28E-07	9.67E-06
I-129	5.92E-01	2.01E-01	3.90E-01	5.55E-06	2.62E-05
Cs-134	1.60E-01	5.35E-02	9.07E-02	1.48E-06	1.61E-02
Cs-135	1.46E-02	5.16E-03	9.42E-03	1.46E-07	8.39E-08
Cs-137+D	1.09E-01	3.65E-02	6.63E-02	1.02E-06	6.56E-03
Ba-133	6.92E-03	2.48E-03	5.00E-04	2.50E-07	3.94E-03
Pm-147	9.57E-04	7.67E-04	1.89E-04	1.25E-06	9.89E-08
Sm-147	1.73E-01	1.35E-01	3.51E-02	2.39E-03	0.00
Sm-151	3.59E-04	2.84E-04	7.38E-05	9.60E-07	2.15E-09

Table B-9. Low Water Infiltration Case: HSRAM Agricultural.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Eu-150	2.28E-02	4.64E-03	1.20E-03	8.57E-06	1.70E-02
Eu-152	1.89E-02	4.73E-03	1.22E-03	7.07E-06	1.29E-02
Eu-154	2.28E-02	6.97E-03	1.79E-03	9.14E-06	1.40E-02
Eu-155	1.78E-03	1.12E-03	2.84E-04	1.32E-06	3.77E-04
Gd-152	1.52E-01	1.18E-01	2.65E-02	7.78E-03	0.00
Ho-166m	2.74E-02	5.89E-03	1.46E-03	2.47E-05	2.01E-02
Re-187	1.13E-05	6.94E-06	4.39E-06	1.74E-09	0.00
Tl-204	5.72E-03	2.45E-03	3.26E-03	7.70E-08	8.22E-06
Pb-205	1.42E-03	1.19E-03	2.28E-04	1.25E-07	1.55E-08
Pb-210+D	4.72E+00	3.92E+00	8.00E-01	4.46E-04	1.29E-05
Bi-207	2.26E-02	4.00E-03	9.31E-04	6.40E-07	1.76E-02
Po-209	2.12E+00	1.74E+00	3.80E-01	3.78E-04	3.86E-05
Po-210	1.64E+00	1.39E+00	2.52E-01	2.96E-04	5.39E-08
Ra-226+D	1.21E+00	9.71E-01	2.17E-01	2.75E-04	2.07E-02
Ra-228+D	1.30E+00	1.05E+00	2.31E-01	2.05E-04	1.31E-02
Ac-227+D	1.26E+01	1.08E+01	1.56E+00	2.15E-01	4.10E-03
Th-228+D	7.00E-01	5.92E-01	8.15E-02	1.10E-02	1.58E-02
Th-229+D	3.43E+00	2.94E+00	4.17E-01	6.91E-02	3.22E-03
Th-230	4.67E-01	4.00E-01	5.67E-02	1.04E-02	6.10E-06
Th-232	2.33E+00	1.99E+00	2.86E-01	5.25E-02	5.91E-04
Pa-231	8.88E+00	7.74E+00	1.10E+00	4.10E-02	4.45E-04
U-232	1.23E+00	9.56E-01	2.52E-01	2.11E-02	2.31E-03
U-233	2.71E-01	2.11E-01	5.53E-02	4.32E-03	3.07E-06
U-234	2.65E-01	2.07E-01	5.41E-02	4.22E-03	8.72E-07
U-235+D	2.52E-01	1.95E-01	5.10E-02	3.94E-03	1.61E-03
U-236	2.52E-01	1.96E-01	5.14E-02	4.00E-03	4.64E-07
U-238+D	2.51E-01	1.96E-01	5.12E-02	3.78E-03	2.52E-04
Np-237+D	3.72E+00	3.24E+00	4.64E-01	1.73E-02	2.24E-03
Pu-236	9.77E-01	8.54E-01	1.18E-01	4.63E-03	6.45E-06
Pu-238	2.68E+00	2.34E+00	3.28E-01	1.25E-02	3.30E-07
Pu-239	2.96E+00	2.58E+00	3.64E-01	1.37E-02	6.23E-07
Pu-240	2.96E+00	2.58E+00	3.64E-01	1.37E-02	3.21E-07
Pu-241+D	5.73E-02	5.00E-02	7.05E-03	2.64E-04	9.84E-08
Pu-242	2.81E+00	2.45E+00	3.45E-01	1.32E-02	2.81E-07
Pu-244+D	2.78E+00	2.42E+00	3.41E-01	1.29E-02	3.91E-03
Am-241	3.05E+00	2.66E+00	3.74E-01	1.42E-02	9.60E-05
Am-242m+D	2.95E+00	2.57E+00	3.63E-01	1.36E-02	1.42E-04
Am-243+D	3.04E+00	2.65E+00	3.73E-01	1.41E-02	1.91E-03
Cm-242	9.54E-02	8.40E-02	1.09E-02	5.47E-04	2.18E-07
Cm-243	2.11E+00	1.83E+00	2.63E-01	9.82E-03	1.23E-03
Cm-244	1.69E+00	1.47E+00	2.12E-01	7.93E-03	2.72E-07
Cm-245	3.14E+00	2.73E+00	3.93E-01	1.46E-02	7.40E-04

Table B-9. Low Water Infiltration Case: HSRAM Agricultural.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Cm-246	3.10E+00	2.70E+00	3.89E-01	1.44E-02	2.55E-07
Cm-247+D	2.87E+00	2.50E+00	3.60E-01	1.32E-02	3.78E-03
Cm-248	1.14E+01	9.93E+00	1.43E+00	5.28E-02	1.92E-07
Cm-250+D	6.52E+01	5.67E+01	8.17E+00	3.01E-01	3.70E-03
Bk-247	3.93E+00	3.43E+00	4.83E-01	1.84E-02	9.25E-04
Cf-248	3.01E-01	2.44E-01	5.51E-02	1.61E-03	2.10E-07
Cf-249	4.37E+00	3.46E+00	8.90E-01	1.85E-02	3.76E-03
Cf-250	1.96E+00	1.55E+00	3.96E-01	8.38E-03	2.55E-07
Cf-251	4.47E+00	3.54E+00	9.11E-01	1.88E-02	1.13E-03
Cf-252	9.86E-01	7.88E-01	1.93E-01	5.01E-03	3.49E-07

Table B-10. Low Water Infiltration Case: Native American Subsistence Resident.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
H-3	1.03E-04	7.01E-05	2.79E-05	5.09E-06	0.00
Be-10	8.25E-03	5.10E-03	3.08E-03	6.44E-05	2.32E-06
C-14	1.34E-02	2.29E-03	1.11E-02	3.80E-07	2.91E-08
Na-22	1.52E-01	1.26E-02	1.17E-01	1.39E-06	2.33E-02
Si-32+D	1.95E-02	1.20E-02	7.21E-03	1.87E-04	2.46E-05
Cl-36	1.80E-01	3.32E-03	1.76E-01	3.98E-06	4.77E-06
K-40	8.37E-02	2.04E-02	6.15E-02	2.26E-06	1.86E-03
Ti-44+D	1.50E-01	2.69E-02	9.70E-02	8.23E-05	2.58E-02
V-49	1.09E-04	6.72E-05	4.12E-05	6.25E-08	0.00
Mn-54	1.24E-02	3.03E-03	2.08E-03	1.21E-06	7.28E-03
Fe-55	1.47E-03	6.65E-04	8.01E-04	4.89E-07	0.00
Co-60	1.04E-01	2.95E-02	4.61E-02	3.98E-05	2.82E-02
Ni-59	4.56E-04	2.30E-04	2.25E-04	2.40E-07	0.00
Ni-63	1.25E-03	6.32E-04	6.18E-04	5.64E-07	0.00
Se-79	3.10E-02	9.53E-03	2.15E-02	1.79E-06	4.08E-08
Rb-87	2.33E-02	5.39E-03	1.79E-02	5.88E-07	3.08E-07
Sr-90+D	3.38E-01	1.68E-01	1.71E-01	4.51E-05	4.98E-05
Zr-93	3.29E-03	1.82E-03	1.42E-03	5.84E-05	3.77E-09
Nb-91	7.63E-03	5.72E-04	7.03E-03	5.31E-06	2.48E-05
Nb-93m	7.52E-03	5.72E-04	6.95E-03	5.31E-06	2.24E-07
Nb-94	1.23E-01	7.82E-03	9.62E-02	7.53E-05	1.85E-02
Mo-93	3.22E-03	1.48E-03	1.73E-03	5.17E-06	1.28E-06
Tc-99	1.23E-02	1.60E-03	1.07E-02	1.51E-06	2.02E-07
Ru-106+D	4.96E-02	3.00E-02	1.76E-02	8.65E-05	1.96E-03
Pd-107	6.34E-04	1.63E-04	4.69E-04	2.33E-06	0.00
Ag-108m+D	6.91E-02	8.34E-03	4.19E-02	5.15E-05	1.88E-02
Cd-109	2.62E-02	1.43E-02	1.18E-02	2.07E-05	2.63E-05
Cd-113m	3.32E-01	1.76E-01	1.55E-01	2.78E-04	1.37E-06
In-115	3.20E-01	1.73E-01	1.46E-01	6.81E-04	8.69E-07
Sn-121m+D	1.03E-02	2.46E-03	7.80E-03	2.17E-06	4.60E-06
Sn-126+D	1.20E-01	2.31E-02	7.34E-02	1.84E-05	2.32E-02
Sb-125	9.78E-03	3.08E-03	2.31E-03	2.22E-06	4.39E-03
Te-125m	6.70E-03	4.02E-03	2.67E-03	1.31E-06	9.67E-06
I-129	1.21E+00	3.02E-01	9.10E-01	3.16E-05	2.62E-05
Cs-134	3.06E-01	8.03E-02	2.10E-01	8.41E-06	1.61E-02
Cs-135	2.94E-02	7.74E-03	2.17E-02	8.28E-07	8.39E-08
Cs-137+D	2.14E-01	5.48E-02	1.53E-01	5.81E-06	6.56E-03
Ba-133	1.02E-02	3.72E-03	2.52E-03	1.42E-06	3.94E-03
Pm-147	1.95E-03	1.15E-03	7.97E-04	7.12E-06	9.89E-08
Sm-147	3.62E-01	2.03E-01	1.45E-01	1.36E-02	0.00
Sm-151	7.37E-04	4.26E-04	3.06E-04	5.46E-06	2.15E-09

Table B-10. Low Water Infiltration Case: Native American Subsistence Resident.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Eu-150	2.90E-02	6.96E-03	4.99E-03	4.88E-05	1.70E-02
Eu-152	2.51E-02	7.10E-03	5.06E-03	4.02E-05	1.29E-02
Eu-154	3.19E-02	1.05E-02	7.43E-03	5.20E-05	1.40E-02
Eu-155	3.24E-03	1.68E-03	1.18E-03	7.53E-06	3.77E-04
Gd-152	3.39E-01	1.76E-01	1.19E-01	4.42E-02	0.00
Ho-166m	3.53E-02	8.84E-03	6.22E-03	1.41E-04	2.01E-02
Re-187	2.42E-05	1.04E-05	1.37E-05	9.90E-09	0.00
Tl-204	1.14E-02	3.68E-03	7.75E-03	4.38E-07	8.22E-06
Pb-205	2.94E-03	1.78E-03	1.16E-03	7.13E-07	1.55E-08
Pb-210+D	9.91E+00	5.88E+00	4.03E+00	2.52E-03	1.29E-05
Bi-207	2.78E-02	6.00E-03	4.22E-03	3.64E-06	1.76E-02
Po-209	4.72E+00	2.61E+00	2.12E+00	2.15E-03	3.86E-05
Po-210	3.55E+00	2.08E+00	1.47E+00	1.70E-03	5.39E-08
Ra-226+D	2.47E+00	1.46E+00	9.88E-01	1.57E-03	2.07E-02
Ra-228+D	2.65E+00	1.58E+00	1.06E+00	1.05E-03	1.31E-02
Ac-227+D	2.66E+01	1.62E+01	9.20E+00	1.22E+00	4.10E-03
Th-228+D	1.45E+00	8.88E-01	4.87E-01	6.27E-02	1.58E-02
Th-229+D	7.30E+00	4.41E+00	2.49E+00	3.93E-01	3.22E-03
Th-230	9.99E-01	6.00E-01	3.39E-01	5.93E-02	6.10E-06
Th-232	4.99E+00	2.99E+00	1.70E+00	2.98E-01	5.91E-04
Pa-231	1.84E+01	1.16E+01	6.59E+00	2.33E-01	4.45E-04
U-232	2.64E+00	1.43E+00	1.08E+00	1.20E-01	2.31E-03
U-233	5.77E-01	3.16E-01	2.36E-01	2.46E-02	3.07E-06
U-234	5.65E-01	3.10E-01	2.31E-01	2.40E-02	8.72E-07
U-235+D	5.34E-01	2.92E-01	2.18E-01	2.24E-02	1.61E-03
U-236	5.37E-01	2.95E-01	2.20E-01	2.27E-02	4.64E-07
U-238+D	5.34E-01	2.93E-01	2.19E-01	2.15E-02	2.52E-04
Np-237+D	7.73E+00	4.86E+00	2.76E+00	9.83E-02	2.24E-03
Pu-236	2.02E+00	1.28E+00	7.10E-01	2.64E-02	6.45E-06
Pu-238	5.55E+00	3.50E+00	1.98E+00	7.13E-02	3.30E-07
Pu-239	6.14E+00	3.88E+00	2.19E+00	7.81E-02	6.23E-07
Pu-240	6.14E+00	3.88E+00	2.19E+00	7.81E-02	3.21E-07
Pu-241+D	1.19E-01	7.50E-02	4.24E-02	1.50E-03	9.84E-08
Pu-242	5.83E+00	3.68E+00	2.08E+00	7.48E-02	2.81E-07
Pu-244+D	5.76E+00	3.64E+00	2.05E+00	7.33E-02	3.91E-03
Am-241	6.32E+00	3.99E+00	2.25E+00	8.08E-02	9.60E-05
Am-242m+D	6.11E+00	3.85E+00	2.18E+00	7.75E-02	1.42E-04
Am-243+D	6.30E+00	3.97E+00	2.24E+00	8.01E-02	1.91E-03
Cm-242	1.94E-01	1.26E-01	6.45E-02	3.13E-03	2.18E-07
Cm-243	4.36E+00	2.75E+00	1.56E+00	5.59E-02	1.23E-03
Cm-244	3.51E+00	2.21E+00	1.25E+00	4.51E-02	2.72E-07
Cm-245	6.51E+00	4.10E+00	2.33E+00	8.28E-02	7.40E-04

Table B-10. Low Water Infiltration Case: Native American Subsistence Resident.
Units are mrem per pCi/L in the ground water.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Cm-246	6.44E+00	4.05E+00	2.30E+00	8.21E-02	2.55E-07
Cm-247+D	5.95E+00	3.74E+00	2.13E+00	7.53E-02	3.78E-03
Cm-248	2.37E+01	1.49E+01	8.46E+00	3.00E-01	1.92E-07
Cm-250+D	1.35E+02	8.51E+01	4.84E+01	1.71E+00	3.70E-03
Bk-247	8.16E+00	5.15E+00	2.90E+00	1.04E-01	9.25E-04
Cf-248	6.16E-01	3.66E-01	2.41E-01	9.20E-03	2.10E-07
Cf-249	9.01E+00	5.19E+00	3.71E+00	1.05E-01	3.76E-03
Cf-250	4.04E+00	2.33E+00	1.66E+00	4.77E-02	2.55E-07
Cf-251	9.22E+00	5.31E+00	3.80E+00	1.07E-01	1.13E-03
Cf-252	2.03E+00	1.18E+00	8.17E-01	2.85E-02	3.49E-07

Table B-11. Low Water Infiltration Case: Columbia River Population.
Units are person-rem per pCi/L in the Columbia River.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
H-3	2.29E-01	1.73E-01	3.48E-02	2.09E-02	0.00
Be-10	1.52E+01	1.26E+01	2.48E+00	1.50E-01	5.60E-03
C-14	1.95E+01	5.64E+00	1.39E+01	8.87E-04	7.02E-05
Na-22	2.14E+02	3.11E+01	1.26E+02	3.25E-03	5.63E+01
Si-32+D	3.55E+01	2.97E+01	5.33E+00	4.37E-01	5.92E-02
Cl-36	1.88E+02	8.18E+00	1.80E+02	9.29E-03	1.15E-02
K-40	1.22E+02	5.02E+01	6.72E+01	5.26E-03	4.49E+00
Ti-44+D	2.40E+02	6.64E+01	1.12E+02	1.92E-01	6.23E+01
V-49	2.03E-01	1.66E-01	3.66E-02	1.46E-04	0.00
Mn-54	2.66E+01	7.48E+00	1.59E+00	2.84E-03	1.76E+01
Fe-55	2.72E+00	1.64E+00	1.08E+00	1.14E-03	0.00
Co-60	1.98E+02	7.26E+01	5.72E+01	9.29E-02	6.81E+01
Ni-59	7.93E-01	5.67E-01	2.26E-01	5.60E-04	0.00
Ni-63	2.18E+00	1.56E+00	6.20E-01	1.32E-03	0.00
Se-79	5.03E+01	2.35E+01	2.68E+01	4.18E-03	9.85E-05
Rb-87	3.23E+01	1.33E+01	1.90E+01	1.37E-03	7.43E-04
Sr-90+D	5.53E+02	4.13E+02	1.39E+02	1.05E-01	1.20E-01
Zr-93	6.10E+00	4.48E+00	1.48E+00	1.36E-01	9.10E-06
Nb-91	1.23E+01	1.41E+00	1.08E+01	1.24E-02	5.98E-02
Nb-93m	1.21E+01	1.41E+00	1.07E+01	1.24E-02	5.42E-04
Nb-94	2.13E+02	1.93E+01	1.48E+02	1.76E-01	4.47E+01
Mo-93	5.46E+00	3.65E+00	1.80E+00	1.21E-02	3.10E-03
Tc-99	1.46E+01	3.94E+00	1.07E+01	3.53E-03	4.87E-04
Ru-106+D	9.36E+01	7.40E+01	1.47E+01	2.02E-01	4.72E+00
Pd-107	8.43E-01	4.02E-01	4.35E-01	5.43E-03	0.00
Ag-108m+D	1.05E+02	2.06E+01	3.87E+01	1.20E-01	4.55E+01
Cd-109	4.52E+01	3.54E+01	9.74E+00	4.83E-02	6.34E-02
Cd-113m	5.63E+02	4.35E+02	1.28E+02	6.49E-01	3.31E-03
In-115	5.91E+02	4.27E+02	1.63E+02	1.59E+00	2.10E-03
Sn-121m+D	1.91E+01	6.08E+00	1.31E+01	5.05E-03	1.11E-02
Sn-126+D	2.36E+02	5.70E+01	1.23E+02	4.29E-02	5.59E+01

Table B-11. Low Water Infiltration Case: Columbia River Population.
Units are person-rem per pCi/L in the Columbia River.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
Sb-125	2.03E+01	7.59E+00	2.15E+00	5.18E-03	1.06E+01
Te-125m	1.30E+01	9.91E+00	3.11E+00	3.07E-03	2.33E-02
I-129	1.64E+03	7.45E+02	8.91E+02	7.38E-02	6.32E-02
Cs-134	4.75E+02	1.98E+02	2.38E+02	1.96E-02	3.89E+01
Cs-135	4.39E+01	1.91E+01	2.48E+01	1.93E-03	2.02E-04
Cs-137+D	3.25E+02	1.35E+02	1.74E+02	1.35E-02	1.58E+01
Ba-133	2.07E+01	9.18E+00	2.06E+00	3.31E-03	9.51E+00
Pm-147	3.63E+00	2.84E+00	7.76E-01	1.66E-02	2.39E-04
Sm-147	6.75E+02	5.00E+02	1.43E+02	3.17E+01	0.00
Sm-151	1.36E+00	1.05E+00	3.01E-01	1.27E-02	5.19E-06
Eu-150	6.32E+01	1.72E+01	4.91E+00	1.14E-01	4.10E+01
Eu-152	5.38E+01	1.75E+01	4.98E+00	9.38E-02	3.12E+01
Eu-154	6.70E+01	2.58E+01	7.29E+00	1.21E-01	3.38E+01
Eu-155	6.21E+00	4.13E+00	1.15E+00	1.76E-02	9.11E-01
Gd-152	6.47E+02	4.35E+02	1.09E+02	1.03E+02	0.00
Ho-166m	7.65E+01	2.18E+01	6.00E+00	3.28E-01	4.84E+01
Re-187	3.97E-02	2.57E-02	1.40E-02	2.31E-05	0.00
Tl-204	2.02E+01	9.07E+00	1.11E+01	1.02E-03	1.98E-02
Pb-205	5.33E+00	4.40E+00	9.28E-01	1.66E-03	3.74E-05
Pb-210+D	1.78E+04	1.45E+04	3.25E+03	5.87E+00	3.10E-02
Bi-207	6.07E+01	1.48E+01	3.43E+00	8.49E-03	4.25E+01
Po-209	8.56E+03	6.43E+03	2.13E+03	5.01E+00	9.31E-02
Po-210	6.63E+03	5.13E+03	1.50E+03	3.97E+00	1.30E-04
Ra-226+D	4.41E+03	3.59E+03	7.64E+02	3.65E+00	4.99E+01
Ra-228+D	4.74E+03	3.89E+03	8.16E+02	2.34E+00	3.15E+01
Ac-227+D	4.96E+04	4.00E+04	6.74E+03	2.85E+03	9.90E+00
Th-228+D	2.73E+03	2.19E+03	3.56E+02	1.46E+02	3.82E+01
Th-229+D	1.36E+04	1.09E+04	1.82E+03	9.17E+02	7.78E+00
Th-230	1.87E+03	1.48E+03	2.48E+02	1.38E+02	1.47E-02
Th-232	9.31E+03	7.37E+03	1.24E+03	6.96E+02	1.42E+00
Pa-231	3.40E+04	2.86E+04	4.81E+03	5.44E+02	1.07E+00

Table B-11. Low Water Infiltration Case: Columbia River Population.
Units are person-rem per pCi/L in the Columbia River.

Nuclide	Total	Ingest (Drink)	Ingest (Other)	Inhale	External
U-232	4.74E+03	3.54E+03	9.16E+02	2.80E+02	5.58E+00
U-233	1.04E+03	7.80E+02	2.00E+02	5.73E+01	7.40E-03
U-234	1.02E+03	7.64E+02	1.96E+02	5.60E+01	2.11E-03
U-235+D	9.62E+02	7.21E+02	1.85E+02	5.22E+01	3.88E+00
U-236	9.65E+02	7.26E+02	1.86E+02	5.30E+01	1.12E-03
U-238+D	9.60E+02	7.24E+02	1.85E+02	5.01E+01	6.09E-01
Np-237+D	1.42E+04	1.20E+04	2.03E+03	2.29E+02	5.41E+00
Pu-236	3.74E+03	3.16E+03	5.18E+02	6.15E+01	1.56E-02
Pu-238	1.02E+04	8.64E+03	1.44E+03	1.66E+02	7.96E-04
Pu-239	1.13E+04	9.56E+03	1.60E+03	1.82E+02	1.50E-03
Pu-240	1.13E+04	9.56E+03	1.60E+03	1.82E+02	7.75E-04
Pu-241+D	2.19E+02	1.85E+02	3.10E+01	3.50E+00	2.38E-04
Pu-242	1.08E+04	9.07E+03	1.51E+03	1.74E+02	6.77E-04
Pu-244+D	1.06E+04	8.96E+03	1.50E+03	1.71E+02	9.43E+00
Am-241	1.17E+04	9.83E+03	1.64E+03	1.88E+02	2.32E-01
Am-242m+D	1.13E+04	9.50E+03	1.59E+03	1.81E+02	3.42E-01
Am-243+D	1.16E+04	9.80E+03	1.64E+03	1.87E+02	4.61E+00
Cm-242	3.65E+02	3.11E+02	4.73E+01	7.31E+00	5.25E-04
Cm-243	8.05E+03	6.78E+03	1.14E+03	1.30E+02	2.96E+00
Cm-244	6.48E+03	5.45E+03	9.16E+02	1.05E+02	6.57E-04
Cm-245	1.20E+04	1.01E+04	1.70E+03	1.93E+02	1.78E+00
Cm-246	1.19E+04	9.99E+03	1.68E+03	1.91E+02	6.15E-04
Cm-247+D	1.10E+04	9.23E+03	1.56E+03	1.76E+02	9.12E+00
Cm-248	4.36E+04	3.67E+04	6.19E+03	7.00E+02	4.64E-04
Cm-250+D	2.49E+05	2.10E+05	3.54E+04	3.99E+03	8.92E+00
Bk-247	1.51E+04	1.27E+04	2.12E+03	2.44E+02	2.23E+00
Cf-248	1.15E+03	9.02E+02	2.28E+02	2.15E+01	5.07E-04
Cf-249	1.67E+04	1.28E+04	3.66E+03	2.45E+02	9.08E+00
Cf-250	7.49E+03	5.75E+03	1.63E+03	1.11E+02	6.14E-04
Cf-251	1.71E+04	1.31E+04	3.74E+03	2.50E+02	2.73E+00

Table B-12. Ratio of Total Dose to Drinking Water Dose.

Nuclide	Residential	Agricultural	All Pathways	NASR	Population
H-3	1.05	1.23	1.32	1.47	1.32
Be-10	1.14	1.17	1.27	1.62	1.21
C-14	1.21	4.22	4.14	5.87	3.46
Na-22	3.92	10.51	8.39	12.11	6.88
Si-32+D	1.15	1.16	1.25	1.62	1.20
Cl-36	7.42	26.27	29.41	54.13	22.98
K-40	1.33	3.01	2.82	4.11	2.43
Ti-44+D	2.58	4.89	4.29	5.56	3.62
V-49	1.13	1.18	1.29	1.61	1.22
Mn-54	4.75	4.81	4.14	4.09	3.56
Fe-55	1.14	1.63	1.84	2.21	1.66
Co-60	2.58	3.35	3.15	3.52	2.73
Ni-59	1.14	1.45	1.51	1.98	1.40
Ni-63	1.14	1.45	1.51	1.98	1.40
Se-79	1.14	2.27	2.45	3.25	2.14
Rb-87	1.15	3.17	2.82	4.32	2.43
Sr-90+D	1.16	1.43	1.43	2.02	1.34
Zr-93	1.15	1.32	1.46	1.81	1.36
Nb-91	1.21	10.26	10.86	13.35	8.75
Nb-93m	1.14	10.08	10.68	13.16	8.61
Nb-94	4.70	13.76	13.64	15.68	11.03
Mo-93	1.15	1.58	1.64	2.18	1.50
Tc-99	1.23	5.47	4.49	7.66	3.71
Ru-106+D	1.23	1.27	1.34	1.65	1.26
Pd-107	1.15	2.88	2.40	3.89	2.10
Ag-108m+D	4.54	7.86	6.10	8.29	5.10
Cd-109	1.15	1.32	1.36	1.83	1.28
Cd-113m	1.15	1.35	1.38	1.88	1.30
In-115	1.14	1.35	1.49	1.85	1.39
Sn-121m+D	1.14	3.16	3.74	4.17	3.15
Sn-126+D	2.65	4.67	4.94	5.18	4.14

Table B-12. Ratio of Total Dose to Drinking Water Dose.

Nuclide	Residential	Agricultural	All Pathways	NASR	Population
Sb-125	3.29	3.40	3.07	3.18	2.68
Te-125m	1.11	1.23	1.40	1.67	1.32
I-129	1.15	2.94	2.53	4.01	2.20
Cs-134	1.44	3.00	2.77	3.82	2.40
Cs-135	1.15	2.83	2.65	3.80	2.30
Cs-137+D	1.33	3.00	2.79	3.91	2.41
Ba-133	2.73	2.79	2.55	2.74	2.26
Pm-147	1.14	1.25	1.36	1.70	1.28
Sm-147	1.16	1.28	1.43	1.79	1.35
Sm-151	1.14	1.26	1.38	1.73	1.30
Eu-150	4.80	4.92	4.28	4.16	3.68
Eu-152	3.87	3.99	3.54	3.54	3.07
Eu-154	3.15	3.27	2.96	3.05	2.60
Eu-155	1.48	1.59	1.63	1.93	1.50
Gd-152	1.21	1.29	1.56	1.92	1.49
Ho-166m	4.55	4.66	4.08	3.99	3.51
Re-187	1.17	1.63	1.70	2.32	1.55
Tl-204	1.14	2.33	2.57	3.11	2.23
Pb-205	1.14	1.19	1.27	1.65	1.21
Pb-210+D	1.15	1.20	1.29	1.69	1.22
Bi-207	5.54	5.64	4.80	4.64	4.10
Po-209	1.14	1.22	1.42	1.81	1.33
Po-210	1.12	1.18	1.37	1.71	1.29
Ra-226+D	1.16	1.25	1.29	1.69	1.23
Ra-228+D	1.15	1.23	1.28	1.68	1.22
Ac-227+D	1.16	1.16	1.29	1.64	1.24
Th-228+D	1.18	1.18	1.30	1.64	1.25
Th-229+D	1.17	1.17	1.30	1.66	1.25
Th-230	1.17	1.17	1.31	1.66	1.26
Th-232	1.17	1.17	1.31	1.67	1.26
Pa-231	1.15	1.15	1.24	1.59	1.19

Table B-12. Ratio of Total Dose to Drinking Water Dose.

Nuclide	Residential	Agricultural	All Pathways	NASR	Population
U-232	1.17	1.29	1.41	1.84	1.34
U-233	1.16	1.28	1.40	1.82	1.33
U-234	1.16	1.28	1.40	1.82	1.33
U-235+D	1.17	1.29	1.41	1.83	1.33
U-236	1.16	1.28	1.40	1.82	1.33
U-238+D	1.16	1.28	1.40	1.82	1.33
Np-237+D	1.15	1.15	1.24	1.59	1.19
Pu-236	1.14	1.14	1.23	1.57	1.18
Pu-238	1.15	1.15	1.24	1.58	1.19
Pu-239	1.15	1.15	1.24	1.58	1.19
Pu-240	1.15	1.15	1.24	1.58	1.19
Pu-241+D	1.15	1.15	1.24	1.59	1.19
Pu-242	1.15	1.15	1.24	1.58	1.19
Pu-244+D	1.15	1.15	1.24	1.59	1.19
Am-241	1.15	1.15	1.24	1.58	1.19
Am-242m+D	1.15	1.15	1.24	1.59	1.19
Am-243+D	1.15	1.15	1.24	1.58	1.19
Cm-242	1.13	1.14	1.22	1.54	1.18
Cm-243	1.15	1.15	1.24	1.59	1.19
Cm-244	1.15	1.15	1.24	1.59	1.19
Cm-245	1.15	1.15	1.24	1.59	1.19
Cm-246	1.15	1.15	1.24	1.59	1.19
Cm-247+D	1.15	1.15	1.24	1.59	1.19
Cm-248	1.15	1.15	1.24	1.59	1.19
Cm-250+D	1.15	1.15	1.24	1.59	1.19
Bk-247	1.15	1.15	1.24	1.58	1.19
Cf-248	1.14	1.23	1.35	1.68	1.28
Cf-249	1.15	1.26	1.39	1.74	1.31
Cf-250	1.15	1.26	1.38	1.73	1.30
Cf-251	1.15	1.26	1.39	1.74	1.31
Cf-252	1.14	1.25	1.37	1.71	1.30

Table B-13. Ingestion Dose Factors.
Units are mrem/pCi Ingested.

Nuclide	f1	GENII	EPA	DOE	GENII/EPA	DOE/EPA
H-3	1	6.12E-08	6.40E-08	6.30E-08		
Be-10	0.005	4.70E-06	4.66E-06	4.20E-06		0.90
C-14	1	2.06E-06	2.09E-06	2.10E-06		
Na-22	1	1.06E-05	1.15E-05	1.20E-05	0.92	
Si-32+D	0.01	1.11E-05	1.10E-05	9.40E-06		0.85
Cl-36	1	2.95E-06	3.03E-06	3.00E-06		
K-40	1	1.79E-05	1.86E-05	1.90E-05		
Ti-44+D	0.01	2.35E-05	2.46E-05	1.91E-05		0.78
V-49	0.01	6.04E-08	6.14E-08	5.40E-08		0.88
Mn-54	0.1	2.76E-06	2.77E-06	2.70E-06		
Fe-55	0.1	6.15E-07	6.07E-07	5.80E-07		
Co-60	0.3	2.65E-05	2.69E-05	2.60E-05		
Ni-59	0.05	2.05E-07	2.10E-07	2.00E-07		
Ni-63	0.05	5.72E-07	5.77E-07	5.40E-07		0.94
Se-79	0.8	8.33E-06	8.70E-06	8.30E-06		
Rb-87	1	4.73E-06	4.92E-06	4.80E-06		
Sr-90+D	0.3	1.31E-04	1.53E-04	1.40E-04	0.86	0.92
Zr-93	0.002	1.64E-06	1.66E-06	1.60E-06		
Nb-91	0.01	5.05E-07	5.22E-07	5.30E-07		
Nb-93m	0.01	5.05E-07	5.22E-07	5.30E-07		
Nb-94	0.01	7.25E-06	7.14E-06	5.10E-06		0.71
Mo-93	0.8	1.21E-06	1.35E-06	1.30E-06	0.90	
Tc-99	0.8	2.23E-06	1.46E-06	1.30E-06	1.53	0.89
Ru-106+D	0.05	2.73E-05	2.74E-05	2.10E-05		0.77
Pd-107	0.005	1.50E-07	1.49E-07	1.40E-07		0.94
Ag-108m+D	0.05	7.58E-06	7.62E-06	7.50E-06		
Cd-109	0.05	1.32E-05	1.31E-05	1.20E-05		0.92
Cd-113m	0.05	1.62E-04	1.61E-04	1.50E-04		0.93
In-115	0.02	8.68E-05	1.58E-04	1.40E-04	0.55	0.89
Sn-121m+D	0.02	2.24E-06	2.25E-06	1.99E-06		0.88
Sn-126+D	0.02	2.08E-05	2.11E-05	1.83E-05		0.87

Table B-13. Ingestion Dose Factors.
Units are mrem/pCi Ingested.

Nuclide	f1	GENII	EPA	DOE	GENII/EPA	DOE/EPA
Sb-125+D	0.1	2.83E-06	2.81E-06	2.60E-06		0.93
Te-125m	0.2	3.72E-06	3.67E-06	3.40E-06		0.93
I-129	1	2.49E-04	2.76E-04	2.80E-04	0.90	
Cs-134	1	6.82E-05	7.33E-05	7.40E-05	0.93	
Cs-135	1	6.86E-06	7.07E-06	7.10E-06		
Cs-137+D	1	4.74E-05	5.00E-05	5.00E-05	0.95	
Ba-133	0.1	3.05E-06	3.40E-06	3.20E-06	0.90	0.94
Pm-147	0.0003	1.06E-06	1.05E-06	9.50E-07		0.90
Sm-147	0.0003	1.86E-04	1.85E-04	1.80E-04		
Sm-151	0.0003	3.87E-07	3.89E-07	3.40E-07		0.87
Eu-150	0.001	6.34E-06	6.36E-06	6.20E-06		
Eu-152	0.001	6.48E-06	6.48E-06	6.00E-06		0.93
Eu-154	0.001	9.61E-06	9.55E-06	9.10E-06		
Eu-155	0.001	1.53E-06	1.53E-06	1.30E-06		0.85
Gd-152	0.0003	1.61E-04	1.61E-04	1.50E-04		0.93
Ho-166m	0.0003	8.13E-06	8.07E-06	7.80E-06		
Re-187	0.8	1.45E-08	9.51E-09	8.30E-09	1.52	0.87
Tl-204	1	3.46E-06	3.36E-06	3.20E-06		
Pb-205	0.2	1.64E-06	1.63E-06	1.50E-06		0.92
Pb-210+D	0.2	5.40E-03	5.37E-03	5.11E-03		
Bi-207	0.05	5.49E-06	5.48E-06	4.90E-06		0.89
Po-209	0.1	2.39E-03	2.38E-03	2.00E-03		0.84
Po-210	0.1	1.90E-03	1.90E-03	1.60E-03		0.84
Ra-226+D	0.2	9.51E-04	1.33E-03	1.10E-03	0.72	0.83
Ra-228+D	0.2	8.44E-04	1.44E-03	1.20E-03	0.59	0.83
Ac-227+D	0.001	1.44E-02	1.48E-02	1.46E-02		
Th-228+D	0.0002	5.79E-04	8.11E-04	7.54E-04	0.71	0.93
Th-229+D	0.0002	3.87E-03	4.03E-03	3.91E-03		
Th-230	0.0002	5.48E-04	5.48E-04	5.30E-04		
Th-232	0.0002	2.73E-03	2.73E-03	2.80E-03		
Pa-231	0.001	1.06E-02	1.06E-02	1.10E-02		

Table B-13. Ingestion Dose Factors.
Units are mrem/pCi Ingested.

Nuclide	f1	GENII	EPA	DOE	GENII/EPA	DOE/EPA
U-232	0.05	1.31E-03	1.31E-03	1.30E-03		
U-233	0.05	2.90E-04	2.89E-04	2.70E-04		0.93
U-234	0.05	2.84E-04	2.83E-04	2.60E-04		0.92
U-235+D	0.05	2.67E-04	2.67E-04	2.51E-04		0.94
U-236	0.05	2.69E-04	2.69E-04	2.50E-04		0.93
U-238+D	0.05	2.70E-04	2.68E-04	2.43E-04		0.91
Np-237+D	0.001	5.22E-03	4.44E-03	3.90E-03	1.18	0.88
Pu-236	0.001	1.16E-03	1.17E-03	1.30E-03		1.11
Pu-238	0.001	3.19E-03	3.20E-03	3.80E-03		1.19
Pu-239	0.001	3.53E-03	3.54E-03	4.30E-03		1.21
Pu-240	0.001	3.53E-03	3.54E-03	4.30E-03		1.21
Pu-241+D	0.001	6.79E-05	6.85E-05	8.60E-05		1.26
Pu-242	0.001	3.35E-03	3.36E-03	4.10E-03		1.22
Pu-244+D	0.001	3.32E-03	3.32E-03	4.00E-03		1.20
Am-241	0.001	3.62E-03	3.64E-03	4.50E-03		1.24
Am-242m+D	0.001	3.50E-03	3.52E-03	4.20E-03		1.19
Am-243+D	0.001	3.62E-03	3.63E-03	4.50E-03		1.24
Cm-242	0.001	1.15E-04	1.15E-04	1.10E-04		
Cm-243	0.001	2.50E-03	2.51E-03	2.90E-03		1.16
Cm-244	0.001	2.01E-03	2.02E-03	2.30E-03		1.14
Cm-245	0.001	3.73E-03	3.74E-03	4.50E-03		1.20
Cm-246	0.001	3.70E-03	3.70E-03	4.50E-03		1.22
Cm-247+D	0.001	3.40E-03	3.42E-03	4.10E-03		1.20
Cm-248	0.001	1.36E-02	1.36E-02	1.60E-02		1.18
Cm-250+D	0.001	7.76E-02	7.77E-02	7.77E-02		
Bk-247	0.001	3.81E-03	4.70E-03	2.30E-03	0.81	0.49
Cf-248	0.001	3.39E-04	3.34E-04	2.80E-04		0.84
Cf-249	0.001	4.75E-03	4.74E-03	4.60E-03		
Cf-250	0.001	2.13E-03	2.13E-03	1.90E-03		0.89
Cf-251	0.001	4.82E-03	4.85E-03	4.60E-03		0.95
Cf-252	0.001	1.09E-03	1.08E-03	9.40E-04		0.87

Table B-13. Ingestion Dose Factors.
Units are mrem/pCi Ingested.

Nuclide	f1	GENII	EPA	DOE	GENII/EPA	DOE/EPA

Notes:

(1) GENII Internal DF are from the July 1999 library revision by PDR. EPA Inhalation & Ingestion dose factors from Federal Guidance Report Number 11, EPA-520/1-88-020, Sept 1988. DOE Ingestion & Inhalation dose factors from DOE/EH-0071, (DE88-014297), July 1988. All are 50 year committed EDE.

(2) "DF" means dose factor. "f1" is the fraction of ingested activity reaching body fluids.

(3) The short-lived radioactive progeny shown on Table B-1 are assumed to be in secular equilibrium with their parent nuclide. The dose factors for implicit daughters have been added to the parent dose factor to give the values shown.

(4) The last two columns show ratios of GENII and DOE ingestion dose factors to the EPA dose factors. Ratios of dose factors within 5% of the EPA value are not shown.

Table B-14. Inhalation Dose Factors
Units are mrem/pCi Inhaled.

Nuclide	Lung Model	GENII	EPA	DOE	GENII/EPA	DOE/EPA
H-3	H2O	9.02E-08	9.60E-08	9.45E-08	0.94	
Be-10	Y	3.54E-04	3.54E-04	3.50E-04		
C-14	Organic	2.06E-06	2.09E-06	2.10E-06		
Na-22	D	7.11E-06	7.66E-06	8.00E-06	0.93	
Si-32+D	Y	1.02E-03	1.03E-03	1.01E-03		
Cl-36	W	2.21E-05	2.19E-05	2.00E-05		0.91
K-40	D	1.19E-05	1.24E-05	1.20E-05		
Ti-44+D	D	4.18E-04	4.52E-04	4.50E-04	0.92	
V-49	W	3.46E-07	3.45E-07	2.80E-07		0.81
Mn-54	W	6.36E-06	6.70E-06	6.40E-06	0.95	
Fe-55	D	2.74E-06	2.69E-06	2.60E-06		
Co-60	Y	2.00E-04	2.19E-04	1.50E-04	0.91	0.68
Ni-59	D	1.28E-06	1.32E-06	1.30E-06		
Ni-63	D	3.06E-06	3.10E-06	3.00E-06		
Se-79	W	9.49E-06	9.84E-06	8.90E-06		0.9
Rb-87	D	3.18E-06	3.23E-06	3.30E-06		
Sr-90+D	D	2.10E-04	2.48E-04	2.37E-04	0.85	
Zr-93	D	3.16E-04	3.21E-04	3.20E-04		
Nb-91	Y	2.94E-05	2.92E-05	2.80E-05		
Nb-93m	Y	2.94E-05	2.92E-05	2.80E-05		
Nb-94	Y	3.91E-04	4.14E-04	3.30E-04	0.94	0.8
Mo-93	Y	2.80E-05	2.84E-05	2.80E-05		
Tc-99	W	9.00E-06	8.33E-06	7.50E-06	1.08	0.9
Ru-106+D	Y	4.75E-04	4.77E-04	4.40E-04		0.92
Pd-107	Y	1.29E-05	1.28E-05	1.30E-05		
Ag-108m+D	Y	2.58E-04	2.83E-04	2.00E-04	0.91	0.71
Cd-109	D	1.15E-04	1.14E-04	1.00E-04		0.88
Cd-113m	D	1.54E-03	1.53E-03	1.40E-03		0.92
In-115	D	2.02E-03	3.74E-03	3.40E-03	0.54	0.91
Sn-121m+D	W	1.18E-05	1.19E-05	9.26E-06		0.78
Sn-126+D	W	1.00E-04	1.01E-04	7.54E-05		0.75
Sb-125+D	W	1.23E-05	1.22E-05	9.80E-06		0.8
Te-125m	W	7.18E-06	7.29E-06	6.70E-06		0.92
I-129	D	1.51E-04	1.74E-04	1.80E-04	0.87	
Cs-134	D	4.28E-05	4.63E-05	4.70E-05	0.92	
Cs-135	D	4.49E-06	4.55E-06	4.50E-06		
Cs-137+D	D	2.98E-05	3.19E-05	3.20E-05	0.93	
Ba-133	D	6.00E-06	7.81E-06	6.90E-06	0.77	0.88
Pm-147	Y	3.92E-05	3.92E-05	3.40E-05		0.87

Table B-14. Inhalation Dose Factors
Units are mrem/pCi Inhaled.

Nuclide	Lung Model	GENII	EPA	DOE	GENII/EPA	DOE/EPA
Sm-147	W	7.48E-02	7.47E-02	7.10E-02		
Sm-151	W	3.01E-05	3.00E-05	2.90E-05		
Eu-150	W	2.50E-04	2.68E-04	2.70E-04	0.93	
Eu-152	W	2.11E-04	2.21E-04	2.20E-04		
Eu-154	W	2.78E-04	2.86E-04	2.60E-04		0.91
Eu-155	W	4.12E-05	4.14E-05	3.90E-05		0.94
Gd-152	D	2.44E-01	2.43E-01	2.40E-01		
Ho-166m	W	7.46E-04	7.73E-04	7.20E-04		0.93
Re-187	W	5.86E-08	5.44E-08	4.90E-08	1.08	0.9
Tl-204	D	2.46E-06	2.41E-06	2.30E-06		
Pb-205	D	3.97E-06	3.92E-06	3.70E-06		0.94
Pb-210+D	D	1.39E-02	1.38E-02	1.32E-02		
Bi-207	W	1.96E-05	2.00E-05	1.40E-05		0.7
Po-209	D	1.19E-02	1.18E-02	1.01E-02		0.86
Po-210	D	9.65E-03	9.40E-03	8.10E-03		0.86
Ra-226+D	W	8.22E-03	8.60E-03	7.91E-03		0.92
Ra-228+D	W	4.40E-03	5.08E-03	4.49E-03	0.87	0.88
Ac-227+D	D	6.71E+00	6.72E+00	6.72E+00		
Th-228+D	W	3.47E-01	3.45E-01	3.13E-01		0.91
Th-229+D	W	2.16E+00	2.16E+00	2.02E+00		0.94
Th-230	W	3.27E-01	3.26E-01	3.20E-01		
Th-232	W	1.64E+00	1.64E+00	1.60E+00		
Pa-231	W	1.29E+00	1.28E+00	1.30E+00		
U-232	Y	6.56E-01	6.59E-01	6.70E-01		
U-233	Y	1.35E-01	1.35E-01	1.30E-01		
U-234	Y	1.32E-01	1.32E-01	1.30E-01		
U-235+D	Y	1.24E-01	1.23E-01	1.20E-01		
U-236	Y	1.26E-01	1.25E-01	1.20E-01		
U-238+D	Y	1.18E-01	1.18E-01	1.20E-01		
Np-237+D	W	6.32E-01	5.40E-01	4.90E-01	1.17	0.91
Pu-236	W	1.45E-01	1.45E-01	1.60E-01		1.1
Pu-238	W	3.90E-01	3.92E-01	4.60E-01		1.17
Pu-239	W	4.30E-01	4.29E-01	5.10E-01		1.19
Pu-240	W	4.30E-01	4.29E-01	5.10E-01		1.19
Pu-241+D	W	8.17E-03	8.25E-03	1.00E-02		1.21
Pu-242	W	4.08E-01	4.11E-01	4.80E-01		1.17
Pu-244+D	W	4.03E-01	4.03E-01	4.80E-01		1.19
Am-241	W	4.41E-01	4.44E-01	5.20E-01		1.17
Am-242m+D	W	4.24E-01	4.26E-01	5.10E-01		1.2
Am-243+D	W	4.41E-01	4.40E-01	5.20E-01		1.18

Table B-14. Inhalation Dose Factors
Units are mrem/pCi Inhaled.

Nuclide	Lung Model	GENII	EPA	DOE	GENII/EPA	DOE/EPA
Cm-242	W	1.75E-02	1.73E-02	1.70E-04		
Cm-243	W	3.07E-01	3.07E-01	3.50E-01		1.14
Cm-244	W	2.48E-01	2.48E-01	2.70E-01		1.09
Cm-245	W	4.55E-01	4.55E-01	5.40E-01		1.19
Cm-246	W	4.51E-01	4.51E-01	5.40E-01		1.2
Cm-247+D	W	4.15E-01	4.14E-01	4.90E-01		1.18
Cm-248	W	1.65E+00	1.65E+00	1.90E+00		1.15
Cm-250+D	W	9.43E+00	9.40E+00	9.40E+00		
Bk-247	W	4.65E-01	5.74E-01	5.50E-01	0.81	
Cf-248	Y	5.11E-02	5.07E-02	4.30E-02		0.85
Cf-249	W	5.77E-01	5.77E-01	5.50E-01		
Cf-250	W	2.63E-01	2.62E-01	2.20E-01		0.84
Cf-251	W	5.87E-01	5.88E-01	5.60E-01		
Cf-252	Y	1.55E-01	1.57E-01	1.30E-01		0.83

Notes:

(1) GENII Internal DF are from the July 1999 library revision by PDR. EPA Inhalation & Ingestion dose factors from Federal Guidance Report Number 11, EPA-520/1-88-020, Sept 1988. DOE Ingestion & Inhalation dose factors from DOE/EH-0071, (DE88-014297), July 1988. All are 50 year committed EDE.

(2) "DF" means dose factor. "Lung" refers to the ICRP lung model classification, "H2O" is tritium vapor (which includes skin absorption), "Organic" means organic carbon, "D" is days, "W" is weeks, and "Y" is years.

(3) The short-lived radioactive progeny shown on Table B-1 are assumed to be in secular equilibrium with their parent nuclide. The dose factors for implicit daughters have been added to the parent dose factor to give the values shown.

(4) The last two columns show ratios of GENII and DOE inhalation dose factors to the EPA dose factors. Ratios of dose factors within 5% of the EPA value are not shown.

Table B-15. External Dose Rate Factors.
Units are mrem/h per Ci/m^2.

Nuclide	GENII	EPA	DOE	GENII/EPA	DOE/EPA
H-3	3.49E-08	0	0	EPA=0	
Be-10	4.33E-01	5.37E-01	0	0.806	DOE=0
C-14	7.51E-03	6.82E-03	0		DOE=0
Na-22	6.75E+03	5.98E+03	2.40E+04	1.13	4.01
Si-32+D	9.62E+00	5.70E+00	0	1.69	DOE=0
Cl-36	8.58E-01	1.16E+00	5.32E-04	0.74	0.000459
K-40	4.87E+02	4.33E+02	1.56E+03	1.12	3.6
Ti-44+D	7.10E+03	6.00E+03	2.57E+04	1.18	4.28
V-49	0	0	8.60E-01		EPA=0
Mn-54	2.53E+03	2.27E+03	9.59E+03	1.11	4.22
Fe-55	1.07E-01	0	2.52E+00	EPA=0	EPA=0
Co-60	7.51E+03	6.87E+03	2.59E+04		3.77
Ni-59	1.31E-01	0	4.75E+00	EPA=0	EPA=0
Ni-63	1.91E-04	0	0	EPA=0	
Se-79	5.37E-03	9.44E-03	0	0.569	DOE=0
Rb-87	4.02E-02	7.13E-02	0	0.564	DOE=0
Sr-90+D	1.97E+01	1.17E+01	0	1.68	DOE=0
Zr-93	1.34E-04	0	0	EPA=0	
Nb-91	5.74E+00	5.74E+00	8.36E+01		14.6
Nb-93m	4.33E-02	5.28E-02	1.17E+01	0.82	222
Nb-94	4.67E+03	4.29E+03	1.81E+04		4.22
Mo-93	2.43E-01	2.99E-01	6.59E+01	0.813	220
Tc-99	5.04E-02	6.35E-02	7.14E-03	0.794	0.112
Ru-106+D	7.32E+02	5.83E+02	2.40E+03	1.26	4.12
Pd-107	4.16E-06	0	0	EPA=0	
Ag-108m+D	5.37E+03	4.37E+03	1.90E+04	1.23	4.35
Cd-109	2.61E+00	7.47E+00	1.08E+02	0.349	14.5
Cd-113m	4.28E-01	3.24E-01	0	1.32	DOE=0
In-115	2.56E-01	2.01E-01	0	1.27	DOE=0
Sn-121m+D	5.15E+00	1.07E+00	0	4.81	DOE=0
Sn-126+D	6.56E+03	5.36E+03	2.37E+04	1.22	4.42
Sb-125+D	1.49E+03	1.12E+03	5.05E+03	1.33	4.51
Te-125m	8.78E+00	7.67E+00	2.40E+02	1.14	31.3
I-129	5.54E+00	6.57E+00	2.51E+02	0.843	38.2
Cs-134	5.23E+03	4.24E+03	1.80E+04	1.23	4.25
Cs-135	1.46E-02	1.94E-02	0	0.753	DOE=0
Cs-137+D	1.82E+03	1.53E+03	6.59E+03	1.19	4.31
Ba-133	1.10E+03	9.36E+02	4.78E+03	1.18	5.11

Table B-15. External Dose Rate Factors.
Units are mrem/h per Ci/m^2.

Nuclide	GENII	EPA	DOE	GENII/EPA	DOE/EPA
Pm-147	2.74E-02	2.53E-02	4.68E-02		1.85
Sm-147	0	0	0		
Sm-151	1.95E-03	4.99E-04	5.93E-02	3.91	119
Eu-150	5.04E+03	3.96E+03	0	1.27	DOE=0
Eu-152	3.60E+03	3.05E+03	1.27E+04	1.18	4.16
Eu-154	3.74E+03	3.34E+03	1.38E+04	1.12	4.13
Eu-155	8.98E+01	9.24E+01	8.16E+02		8.83
Gd-152	0	0	0		
Ho-166m	4.67E+03	4.64E+03	1.88E+04		4.05
Re-187	0	0	0		
Tl-204	1.93E+00	2.04E+00	1.48E+01		7.25
Pb-205	8.75E-02	3.58E-03	8.61E+00	24.4	2410
Pb-210+D	3.85E+00	3.00E+00	3.42E+01	1.28	11.4
Bi-207	4.92E+03	4.11E+03	1.72E+04	1.2	4.18
Po-209	8.95E+00	8.95E+00	4.10E+01		4.58
Po-210	2.68E-02	2.32E-02	9.81E-02	1.16	4.23
Ra-226+D	5.61E+03	4.78E+03	1.92E+04	1.17	4.02
Ra-228+D	3.04E+03	2.62E+03	1.04E+04	1.16	3.97
Ac-227+D	1.08E+03	9.61E+02	5.00E+03	1.12	5.2
Th-228+D	4.92E+03	4.20E+03	1.66E+04	1.17	3.95
Th-229+D	9.04E+02	7.45E+02	4.09E+03	1.21	5.49
Th-230	4.11E-01	6.05E-01	1.03E+01	0.679	17
Th-232	2.13E-01	2.63E-01	7.60E+00	0.81	28.9
Pa-231	9.09E+01	9.11E+01	4.08E+02		4.48
U-232	3.10E-01	4.52E-01	1.17E+01	0.686	25.9
U-233	4.81E-01	6.86E-01	5.70E+00	0.701	8.31
U-234	1.89E-01	2.03E-01	9.21E+00		45.4
U-235+D	2.52E+02	3.74E+02	2.17E+03	0.674	5.8
U-236	9.85E-02	1.08E-01	8.36E+00		77.4
U-238+D	7.10E+01	5.87E+01	2.81E+02	1.21	4.79
Np-237+D	7.13E+02	5.28E+02	3.06E+03	1.35	5.8
Pu-236	9.45E-02	1.14E-01	1.13E+01	0.829	99.1
Pu-238	1.06E-01	7.65E-02	9.79E+00	1.39	128
Pu-239	1.59E-01	1.44E-01	4.31E+00	1.1	29.9
Pu-240	7.29E-02	7.43E-02	9.35E+00		126
Pu-241+D	9.43E-03	9.29E-03	4.40E-02		4.74
Pu-242	9.57E-02	6.49E-02	7.78E+00	1.47	120
Pu-244+D	1.17E+03	9.04E+02	3.86E+03	1.29	4.27
Am-241	1.45E+01	2.22E+01	3.41E+02	0.653	15.4
Am-242m+D	3.58E+01	3.28E+01	2.66E+02		8.11

Table B-15. External Dose Rate Factors.
Units are mrem/h per Ci/m^2.

Nuclide	GENII	EPA	DOE	GENII/EPA	DOE/EPA
Am-243+D	4.49E+02	4.42E+02	2.94E+03		6.65
Cm-242	5.97E-02	8.59E-02	1.07E+01	0.695	125
Cm-243	2.90E+02	2.86E+02	1.67E+03		5.84
Cm-244	5.09E-02	6.39E-02	9.46E+00	0.797	148
Cm-245	1.32E+02	1.71E+02	9.74E+02	0.772	5.7
Cm-246	4.19E-02	5.89E-02	8.37E+00	0.711	142
Cm-247+D	1.30E+03	8.74E+02	4.16E+03	1.49	4.76
Cm-248	3.83E-02	4.45E-02	6.71E+00	0.861	151
Cm-250+D	1.20E+03	8.55E+02	4.40E+03	1.4	5.15
Bk-247	2.32E+02	2.14E+02	0		DOE=0
Cf-248	3.43E-02	6.32E-02	7.68E+00	0.543	122
Cf-249	9.82E+02	8.71E+02	4.02E+03	1.13	4.62
Cf-250	5.37E-02	6.01E-02	7.81E+00	0.894	130
Cf-251	2.40E+02	2.62E+02	1.55E+03		5.92
Cf-252	4.25E-02	8.91E-02	7.23E+00	0.477	81.1

Notes:

(1) GENII external DRF were computed using the EXTDF program. EPA external DRF are from Federal Guidance Report Number 12, EPA 402-R-93-081 (Sept 1993). DOE external DRF are from DOE/EH-0070 (July 1988).

(2) Short-lived radioactive progeny included in the "+D" nuclides are in secular equilibrium with their parent nuclide.

(3) The conversion to area units from volume units assumes a thickness of 0.15 meters. The density correction applied to the EPA (1993) dose rate factors is 1.067. Because Nb-91 and Po-209 are not part of the EPA compilation, the GENII values were used.

(4) The last two columns show ratios of GENII and DOE external dose rate factors to the EPA dose rate factors. Ratios within 10% of the EPA value are not shown.

Appendix C

Equations Used in Major Codes

C. GOVERNING EQUATIONS USED IN MAJOR CODES

C.1 INTRODUCTION

This section briefly summarizes the equations used in the major computer codes. More information can be found in the user's manual for each of the codes:

- CFEST Gupta 1987
- STORM Bacon 2000
- VAM3DF Huyakorn 1999.

C.2 FLOW AND TRANSPORT

C.2.1 Overview

This section discusses the equations important in calculating moisture flow and contaminant transport. All three major codes use the same basic equations. The first section presents the equations on which the water flow calculations are based. The next section discusses contaminant transport associated with moisture flow. The final section describes moisture movement under diffusive conditions.

C.2.2 Moisture Flow

Two distinct moisture content regimes are present during contaminant transport: the unconfined aquifer and the disposal facility/vadose zone. In the unconfined aquifer all the pore spaces are filled with water; that is, the medium is saturated with water. In the disposal facility/ vadose zone the pore spaces between the soil particles are only partially filled with water. These zones are unsaturated.

Water flow through a saturated porous medium, such as the unconfined aquifer, is governed by the empirical relationship known as Darcy's Law (Freeze 1979) and by the conservation of mass. Darcy's law can be expressed as

$$\vec{v} = -\vec{K}\,\vec{\nabla} h \qquad \text{(C.1)}$$

where v is the velocity vector (m/year)
K is the hydraulic conductivity tensor (m/year)
h is the hydraulic head vector (m).

By using the conservation of water entering and leaving a volume and by using Darcy's law to relate the hydraulic gradient to the rate at which water enters and leaves the same volume, transient water flow in a saturated porous media can be expressed as

$$\nabla\left[\overleftrightarrow{K}\nabla h\right] = \frac{Sh}{t} + Q \qquad \text{(C.2)}$$

where S is the specific storage (m^{-1})

Q is the source or sink of moisture (y^{-1}).

Darcy's law defines the discharge of water through a cross section of porous media. However, in contamination transport, the average velocity of water flowing through the medium is important. This is because contaminants that are not geochemically retarded move with the water. The average velocity of the pore water is determined by dividing the velocity of the water by the porosity of the medium. Porosity is defined as the ratio of void space to total volume.

In an unsaturated medium, the pores are not completely filled with water. Capillary forces and the dependence of hydraulic conductivity on moisture content also must be considered. Richards equation (Richards 1931) becomes the governing equation:

$$\nabla[K(\psi)\nabla\psi] + \partial K_{zz}(\psi)/\partial z = (d\theta/d\psi)\,(\partial\psi/\partial t) + Q \qquad \text{(C.3)}$$

where K is again the hydraulic conductivity tensor (m/year), but now depends on the pressure head

ψ is the pressure head (m), which depends on the moisture content

θ is the moisture content (dimensionless)

z is the vertical column of moisture (m).

The relationship between the pressure head and the hydraulic head is simply

$$h = \psi + z. \qquad \text{(C.4)}$$

For the performance assessment calculations, the hydraulic conductivity tensor is reduced to a single function, with

$$K_{ij} = K, \text{ for } i = j \text{ and} \qquad \text{(C.5)}$$

$$K_{ij} = 0, \text{ for } i \neq j.$$

The functional dependence of the pressure head on moisture content and of the hydraulic conductivity on pressure head and ultimately on moisture content is discussed in Section 3.4.3.2.

For extremely dry conditions, vapor diffusion may be important. In such conditions, water does not move as a collective body, but rather as single molecules. Such diffusion can be described by Fick's equation,

$$\frac{dm}{dt} = -Dv\,A\frac{dC}{dx} \qquad \text{(C.6)}$$

where dm/dt is the mass rate of water vapor diffusion (g/year)

D_v is the effective vapor diffusion coefficient (m^2/year)

A is the cross section area (m^2)

C is the water vapor mass concentration in the gas phase (g/m^3)

dC/dx is the water vapor mass concentration gradient (g/m^4).

C.2.3 Advective and Diffusive Transport

The equation for the advective and diffusive transport of contaminants can be viewed as a mass balance on a differential volume. The advective-dispersive equation for solute movement through a porous medium with a constant, steady-state flow velocity was developed (Codell 1982) for the limiting case of unidirectional advective transport with three-dimensional dispersion in a homogeneous, saturated aquifer,

$$n_e\frac{\partial C}{\partial t}+(n-ne)\frac{\partial G}{\partial t}+(1-n)\frac{\partial P}{\partial t}+n_e u\frac{\partial C}{\partial x}=$$

$$n_e\left[E_x\frac{\partial^2 C}{\partial x^2}+E_y\frac{\partial^2 C}{\partial y^2}+E_z\frac{\partial^2 C}{\partial z^2}\right]+$$

$$(n-n_e)\left[E_x^1\frac{\partial^2 G}{\partial x^2}+E_y^1\frac{\partial^2 G}{\partial y^2}+E_z^1\frac{\partial^2 G}{\partial z^2}\right]-$$

$$n_e\lambda C-(1-n)\lambda P-(n-n_e)\lambda G \qquad \text{(C.7)}$$

where C is the dissolved concentration in the liquid phase in voids that are interconnected and allow flow [flowing voids] (g/m^3 or Ci/m^3)

G is the dissolved concentration in the liquid phase in voids that are not interconnected and do not allow flow [non-flowing voids] (g/m^3 or Ci/m^3)

P is the particulate concentrations on the solid phase (g/g or Ci/g)

n is the total porosity (dimensionless)

n_e is the effective porosity (dimensionless)

t is the time (year)

u is the x-component of groundwater or pore water velocity (m/year)

E_i is the dispersion coefficient in the flowing voids in the i-th (where i = x, y, or z) direction (m^2/year)

$E_i^{'}$ is the diffusion coefficient in the non-flowing voids in the i-th direction (m^2/year)

λ is the decay constant [= (ln 2)/half life] (y^{-1}).

Each term in the equation represents some aspect of the solute movement through the porous medium. The first term on the left of the equal sign is the accumulation (storage) of the

solute in the liquid phase in the flowing void. The second term is the accumulation in the liquid phase in the nonflowing void. The third term is the accumulation in the solid phase; and the term to the left of the equal sign is x-direction advective transport in the flowing voids in the liquid phase. The first term to the right of the equal sign represents the dispersive transport in the flowing voids in the liquid phase in each direction. The second term represents the diffusive transport in the nonflowing voids in the liquid phase in each direction. The last terms are the chemical degradation or radioactive decay in the liquid phase in the flowing void, in the solid phase, and in the liquid phase in the nonflowing void respectively.

Using the following assumptions:

- The dissolved concentration in the nonflowing voids (G) equals the dissolved concentration in the flowing voids (C) for each time and position
- The contaminant absorption process can be described by a constant, ($K = \rho^* K_d/(1 - n)$), representing the ratio between the contaminant absorbed to the soil matrix (P) and the contaminant dissolved in solution (C)
- The diffusion in the nonflowing void (E'_i) is comparable with the dispersion in the flowing void (E_i),the equation (D.7) can be simplified to

$$\frac{\partial C}{\partial t} + \frac{u}{R_f}\frac{\partial C}{\partial x} = \frac{D_x}{R_f}\frac{\partial^2 C}{\partial x^2} + \frac{D_y}{R_f}\frac{\partial^2 C}{\partial y^2} + \frac{D_z}{R_f}\frac{\partial^2 C}{\partial z^2} - \lambda C \qquad \text{(C.8)}$$

in which

$$D_i = n\, E_i / n_e \qquad \text{(C.9)}$$

and

$$R_f = n / n_e + (\rho^* K_d) / n_e \qquad \text{(C.10)}$$

and where

D_i is the pseudodispersion coefficient (m^2/year)

R_f is the retardation factor (dimensionless)

ρ is the bulk density (g/m^3)

K_d is the equilibrium (partition or distribution) coefficient (m^3/g).

The retardation factor (R_f) is used as a measure of the mobility of constituents in a porous medium.

By making the following substitutions,

$$u^* = u / R_f \qquad \text{(C.11)}$$

and

$$D_i^* = D_i / R_f,$$

the contaminant transport equation can be written as

$$\frac{\partial C}{\partial t} + u \cdot \frac{\partial C}{\partial x} = D_x \cdot \frac{\partial^2 C}{\partial x^2} + D_y \cdot \frac{\partial^2 C}{\partial y^2} + D_z \cdot \frac{\partial^2 C}{\partial z^2} - \lambda C \qquad \text{(C.12)}$$

The first equation of this section [equation (D.1)] specifically addresses the general conditions for saturated flow and solute movement. However, with the following minor modifications, it also can be applied to the unsaturated zone:

- The porosities (n and n_e) are assumed to be equal to the soil matrix moisture content
- The one-dimensional flow is in the vertical direction.
- For this case, the retardation factor is defined by

$$R_f = 1 + (\rho^* K_d) / \theta \qquad \text{(C.13)}$$

where θ is the moisture content of the partially saturated zone and dispersion is considered only in the flow direction.

C.2.4 Vapor Transport

Some contaminants may move upward from the disposal facility to the surface because they are in the vapor phase. Such movement is governed by Fick's second law,

$$D \frac{\partial^2 C}{\partial z^2} = \frac{\partial C}{\partial t} \qquad \text{(C.14)}$$

where

C	is the concentration (g/m^3 or Ci/m^3)
z	is the distance (m)
t	is time (year)
D	is the diffusion coefficient (m^2/year).

The solution for concentration C in the z direction over time t is given by:

$$\frac{C}{C_0} = erfc\left(\frac{z}{2\sqrt{Dt}}\right) \qquad \text{(C.15)}$$

where erfc is the complimentary error function and C/C_o is the relative concentration. The mass transport across the surface becomes

$$\frac{\partial C}{\partial t} = \frac{C_0}{2\sqrt{\pi DT}} \frac{z}{t} \exp\left(\frac{-z^2}{4Dt}\right) \qquad \text{(C.16)}$$

C.3 WASTE FORM

C.3.1 Mass Conservation of Aqueous Solutes

Based on the continuum theory, the model can be represented by a set of partial differential equations (PDE) that describe the mass conservation of solutes in pore fluids. Assuming that the rock matrix does not deform, the mass conservation law of solutes can be written as

$$\frac{\partial(\phi S_w \rho_w C_i)}{\partial t} = -\vec{\nabla} \cdot \vec{J} + \sum_{j=1}^{N_r} v_{ij} W_j, i = 1,2,... \tag{C.17}$$

where ϕ is the porosity

S_w is the water saturation

ρ_w the density of the aqueous solution

C_i is the molal concentration of solute i

J_i is the flux of species i

v_{ij} is the stoichiometric coefficient of species i in reaction j

N_r is the number of total reactions

W_j is the rate of reaction j.

In general, the reaction rate is a nonlinear function of the concentrations. Through the term W_j, one solute species is nonlinearly coupled with other species. The nonlinear partial differential equation [equation (C.1)] simply states that the time rate of change of aqueous species concentrations consists of two parts; one is the contribution of transport and the other is the contribution of all reactions. Assuming the phenomenon of hydrodynamic dispersion can be represented by a Fickian-type law, the first term on the right-hand side can be expanded into

$$-\vec{\nabla} \cdot \vec{J}_i = -\vec{\nabla} \cdot \left(\phi S_w \rho_w D_i \vec{\nabla} C_i + \vec{u} \phi S_w \rho_w C_i\right) \tag{C.18}$$

where D_i is the dispersion coefficient of i

u is the velocity of pore fluid.

The second term at the right-hand side of equation (C.18) represents all types of reactions, including aqueous speciation, redox reactions, solid precipitation and dissolution, and adsorption. In general, chemical reactions can be treated as either equilibrium-controlled or kinetic-controlled. Several of the available reaction-transport models can only treat equilibrium reactions (Mangold 1991). In this case, the nonlinear equation (C.18) can be transformed into a linear equation, making it numerically easier to solve. However, many reactions are kinetically controlled, especially solid dissolution and precipitation reactions. Consequently, both equilibrium and kinetic reactions are included in the model used here. To emphasize this, the reaction term in equation (C.18) is split into two parts: one represents the contributions from equilibrium reactions, the other represents contributions from kinetic reactions

$$\sum_{j=1}^{N_r} \nu_{ij} W_j = \sum_{j=1}^{Ne} \nu_{ij}^e W_j^e + \sum_{j=1}^{N_k} \nu_{ij}^k W_j^k \tag{C.19}$$

where ν_{ij}^e and ν_{ij}^k are the stoichiometric coefficients in equilibrium reactions and kinetic reactions, respectively, while W_j^e and W_j^k are the rates of equilibrium and kinetic reactions, respectively.

To define the mathematical form of the *W's*, note that for any kinetic reaction *j* involving aqueous and solid species *m* with the form

$$\sum \nu_{mj} B_m = 0 \tag{C.20}$$

using the law of mass-action, with activity corrections, the rate can be expressed as

$$W_j^k = A_j k_j \left[\prod_{\nu_{ij}<0} (\gamma_i C_i)^{-\nu_{ij}} - \prod_{\nu_{ij}>0} \frac{(\gamma_i C_i)^{\nu_{ij}}}{K_j^{eq}} \right] \tag{C.21}$$

where A_j is a factor

k_j is the rate constant

γ_i is the activity coefficient, which is a function of the concentrations of all species.

For aqueous reactions, $A_j = 1$. For solid dissolution and precipitation reactions, A_j is the effective reaction surface in unit volume of the porous medium. If we further assume that all solids are spherical grains or can be represented as equivalent spherical grains with radii of $R_{l(j)}$, and that the effective reaction surface is proportional to water saturation, then

$$W_j^k = 4\pi R_{l(j)}^2 n_{l(j)} S_w k_j \left[\prod_{\nu_{ij}<0} (\gamma_i C_i)^{-\nu_{ij}} - \prod_{\nu_{ij}>0} \frac{(\gamma_i C_i)^{\nu_{ij}}}{K_j^{eq}} \right], \tag{C.22}$$

where n_l is the number of grains in unit volume of porous medium.

Several formulas, such as Davies equation, B-dot equation, and Pitzer's model, can be used to calculate γ_i. The B-dot equation with modifications for neutral species adopted by Wolery (1992) is computationally economic and stable, and can handle moderate salinity; making it the most suitable for reaction-transport problems. Although Pitzer's model can handle high ionic strength, computationally it is not feasible for reaction-transport simulations because of its complicated structure and the lack of Pitzer constants for all but a few aqueous species.

C.3.2 Texture Dynamics

Dissolution and precipitation reactions can change the volume fractions of solids as a function of time. For spherical grains, the change rate is

$$\frac{\partial V_l}{\partial t} = 4\pi R_l^2 n_l \frac{\partial R_l}{\partial t}, \tag{C.23}$$

while

$$\frac{\partial R_l}{\partial t} = \sum_j k_j \left[\prod_{\nu_{lj}<0} (\gamma_i C_i)^{-\nu_{lj}} - \prod_{\nu_{lj}>0} \frac{(\gamma_i C_i)^{\nu_{lj}}}{K_j^{eq}} \right] / \rho_j, \tag{C.24}$$

where V_l is the volume fraction of solid l

R_l is the change rate in radius of solid l

ρ_l is the molar density of l.

Equation (C.24) serves as the bridge between the aqueous phase and solid phases so the whole system is mass-conserved.

C.3.3 Contaminant Release Rate From Glass

The equation for the contaminant release rate from glass is taken as

$$J_i = \nu_i a_{H^+}^{-\eta(T)} k_o e^{\frac{E_a}{RT}} \left[1 - \left(\frac{Q}{K} \right)^{\sigma} \right], \tag{C.25}$$

where J_i is the flux of element i

ν_i is the stoichmetric coefficient of element i

a_{H+} is the activity of H^+

η is the pH power law coefficient

T is the temperature

k_o is the intrinsic rate constant

E_a is the activation energy

R is the gas constant

Q is the ion activity product

K is the pseudoequilibrium constant

σ is the average stoichiometric parameter for the overall reaction.

Appendix D

Detailed Results For Estimated Impacts

D. Detailed Results For Estimated Impacts

D.1 Introduction

This appendix summarizes the detailed results from the waste form, vadose zone, groundwater, and INTEG calculations to estimate the impacts for the different sensitivity cases investigated in this performance assessment. The details for the waste form calculations are provided in Bacon 2000. The details for the near-field, far field, and INTEG results can be found in Finfrock 2000. The details for the groundwater results can be found in Bergeron 2000. This appendix also includes a description of the detailed files that are included with this performance assessment on a compact disc that is available on request.

D.2 Waste Form Calculations

The contaminant fluxes for the base analysis case and all sensitivity cases have been provided and discussed in Section 4.5. Table 4-12 also provides a relative comparison of the results at 1,000 and 10,000 years after facility closure for the technetium flux compared to the base analysis case for all waste form sensitivity cases investigated. For additional details on the waste form calculation results, see Bacon 2000.

For completeness, output files from the waste form calculations that have been processed as input files to the far field VAM3D calculations are provided. See Section D.6 for a description of these files.

D.3 Vadose Zone Calculation

The contaminant fluxes to the aquifer for the base analysis case and the best estimate case have been provided and discussed in Section 4.3. The time dependence of the contaminant flux to the aquifer is shown in Figures 4-3 and 4-10 for the mobile (K_d = 0 mL/g) and less mobile (K_d = 0.6, 4.0 and 10. mL/g). This time dependence is reflected in the time dependence of the "vad-casename.bat" files for each sensitivity case see Section D.6).

D.4 Groundwater Calculations

Discussion of the base case flow and transport calculations of the define (WIF) are provided in Section 4.3 and the associated WIFs are summarized in Table 4.2. Discussion of the sensitivity cases are provided in Section 4.7 and the resulting ratios of the sensitivity case WIFs with the base case WIFs are summarized in Table 4.16.

Additional details of the underlying assumptions and supporting groundwater flow and transport calculations for both the base and sensitivity cases are presented in Bergeron 2000.

D.5 Estimated Impacts

Table D-1 summarizes the results for the estimated impacts to the beta-photon drinking water dose, the alpha concentration, and the all-pathway dose for the all-pathway farmer scenario except where noted. The estimated impacts are associated with a downgradient well located 100 m below the ILAW site except where noted. The estimated impacts are provided for 1,000 and 10,000 years after facility closure. The maximum estimated impacts also are provided. In general the maximum estimated impacts for the alpha concentration and all-pathway dose occurs for the maximum time calculated (typically 20,000 years after facility closure [except for the base analysis case where it is 100,000 years after facility closure]). Also included in Table D-1 are the file names for the INTEG calculations.

Table D-1. Summary of Estimated Impacts for 2001 ILAW PA.

INTEG Filename	Case Designation	Estimated Impact at 1,000 y after Facility Closure (1)			Estimated Impact at10,000 y after Facility Closure (1)			Maximum Estimated Impact (1) [peak time or max time for calculation]		
		B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)
42base	Base analysis case	2.13E-05	1.02E-16	7.78E-05	1.02E-02	3.39E-02	7.00E-02	1.25E-02 [76,500 y]	5.42E-01 [100,000 y]	5.88E-01 [100,000 y]
42break	Best estimate case	4.67E-11	2.25E-22	1.71E-10	3.46E-07	5.01E-08	1.31E-06	1.85E-06 [20,000 y]	9.29E-07 [20,000 y]	7.63E-06 [20,000 y]
Facility Cases										
42break	Full barrier	4.67E-11	2.25E-22	1.71E-10	3.46E-07	5.01E-08	1.31E-06	1.85E-06 [20,000 y]	9.29E-07 [20,000 y]	7.63E-06 [20,000 y]
50break	Full barrier - 50 mm/y	2.09E-09	9.64E-21	7.33E-09	7.92E-07	8.94E-07	3.76E-06	3.03E-06 [20,000 y]	5.43E-06 [20,000 y]	1.63E-05 [20,000 y]
42vert	Vertical barrier	0.00E+00	0.00E+00	0.00E+00	2.03E-07	8.97E-09	7.51E-07	1.2E-06 [20,000 y]	3.14E-07 [20,000 y]	4.68E-06 [20,000 y]
42short	Shorter capillary break	8.53E-10	4.10E-21	3.12E-09	1.05E-06	6.03E-07	4.41E-06	3.94E-06 [20,000 y]	5.24E-06 [20,000 y]	1.94E-05 [20,000 y]
42vault	Concrete vault - base	6.05E-03	9.19E-10	2.21E-02	1.81E-02	6.94E-01	7.38E-01	1.85E-02 [20,000 y]	7.41E-01 [20,000 y]	7.90E-01 [20,000 y]
42all	No sideslope	1.10E-06	5.28E-18	4.01E-06	9.98E-03	8.61E-03	4.48E-02	1.21E-02 [17,400 y]	2.76E-01 [20,000 y]	3.13E-01 [20,000 y]
Waste Form Release										
42for	Forward rate	2.08E-04	9.98E-16	7.59E-04	9.08E-02	3.32E-01	6.54E-01	9.89E-02 [20,000 y]	3.13E+00 [20,000 y]	3.41E+00 [20,000 y]
42noion	No ion exchange	1.68E-05	8.06E-17	6.13E-05	8.41E-03	2.72E-02	5.71E-02	1.04E-02 [20,000 y]	2.91E-01 [20,000 y]	3.21E-01 [20,000 y]
42highion	Forward rate + 5X ion exchange	2.18E-04	1.05E-15	7.97E-04	1.00E-01	3.92E-01	7.46E-01	1.09E-01 [18,700 y]	3.45E+00 [20,000 y]	3.77E+00 [20,000 y]
42nosec	No secondary phase	1.06E-05	5.08E-17	3.86E-05	5.39E-03	1.75E-02	3.67E-02	6.60E-03 [20,000 y]	1.86E-01 [20,000 y]	2.05E-01 [20,000 y]
01	Recharge = 0.1 mm/y	6.95E-12	3.34E-23	2.54E-11	7.16E-07	1.84E-08	2.64E-06	3.23E-06 [20,000 y]	6.30E-07 [20,000 y]	1.24E-05 [20,000 y]

Table D-1. Summary of Estimated Impacts for 2001 ILAW PA.

INTEG Filename	Case Designation	Estimated Impact at 1,000 y after Facility Closure (1)			Estimated Impact at10,000 y after Facility Closure (1)			Maximum Estimated Impact (1) [peak time or max time for calculation]		
		B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)
09	Recharge = 0.9 mm/y	3.76E-09	1.81E-20	1.38E-08	1.41E-04	5.18E-06	5.22E-04	1.81E-04 [20,000 y]	2.57E-04 [20,000 y]	9.10E-04 [20,000 y]
50	Recharge = 50 mm/y	4.00E-2	2.63E-01	3.99E-01	3.76E-02	1.52E+00	1.61E+00	7.62E-02 [300 y]	1.56E+00 [3,400 y]	1.65e+00 [3,400 y]
42sandfill	Sand backfill	4.48E-04	2.15E-15	1.64E-03	1.77E-02	1.76E-01	2.35E-01	1.80E-02 [4,700 y]	6.65E-01 [20,000 y]	7.11E-01 [20,000 y]
42sdiff	10X WF aqueous diffusion	4.03E-06	1.94E-17	1.47E-05	1.20E-04	1.21E-03	1.61E-03	1.31E-04 [20,000 y]	4.52E-03 [20,000 y]	4.89E-03 [20,000 y]
42steel	Steel included	2.18E-05	1.05E-16	7.96E-05	1.01E-02	3.39E-02	6.98E-02	1.49E-02 [20,000 y]	3.75E-01 [20,000 y]	4.02E-01 [20,000 y]
42condlayr	Top conditioning Layer	2.11E-05	1.02E-16	7.72E-05	9.44E-03	3.31E-02	6.66E-02	1.10E-02 [20,000 y]	3.26E-01 [20,000 y]	3.58E-01 [20,000 y]
42altglass	HLP-31 glass	2.57E-03	1.24E-14	9.41E-03	6.77E-01	3.87E+00	6.23E+00	6.78E-01 [9,200 y]	2.40E+01 [20,000 y]	2.56E+01 [20,000 y]
Vadose Zone Cases										
42iso	Isotropic field flow	1.32E-05	6.35E-17	4.83E-05	1.02E-02	1.26E-02	4.96E-02	1.22E-02 @ 20,000 y	3.02E-01 @ 20,000 y	3.39E-01 @ 20,000 y
42sand	All sand	5.37E-06	2.58E-17	1.96E-05	1.00E-02	2.54E-03	3.91E-02	1.24E-02 [52,500 y]	5.36E-01 [100,000 y]	5.85E-01 [100,000 y]
42grav	All gravel	1.04E-04	1.84E-09	3.80E-04	1.05E-02	3.60E-01	3.87E-01	1.24E-02 [52,000 y]	5.43E-01 [100,000 y]	5.92E-01 [100,000 y]
42deep	VZ - 3 m thicker	1.78E-05	8.56E-17	6.51E-05	1.02E-02	3.24E-02	6.86E-02	1.24E-02 [52,000 y]	5.37E-01 [100,000 y]	5.86E-01 [100,000 y]
42dike	Clastic dike	1.91E-05	9.16E-17	6.96E-05	1.01E-02	2.81E-02	6.42E-02	1.22E-02 [20,000 y]	3.32E-01 [20,000 y]	3.68E-01 [20,000 y]
42ukd0	U $K_d = 0$	2.13E-05	6.12E-04	2.09E-04	1.02E-02	3.07E-01	1.29E-01	1.24E-02 [52,500 y]	5.41E-01 [100,000 y]	5.87E-01 [100,000 y]

Table D-1. Summary of Estimated Impacts for 2001 ILAW PA.

INTEG Filename	Case Designation	Estimated Impact at 1,000 y after Facility Closure [1]			Estimated Impact at10,000 y after Facility Closure [1]			Maximum Estimated Impact [1] [peak time or max time for calculation]		
		B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)
42kd0	K_d = 0 for all radionuclides [2]	3.11E-05	1.6E-02 (3.78E-06)	3.68E-02	1.49E-02	4.34e+00 (5.95E-03)	8.69E+00	1.82E-02 [54,500 y]	4.70e+00 [15,000 y] (5.15E-02 [100,000 y])	9.05E+00 [13,000 y]
42diff	10X increase in Difusion	3.17E-05	1.52E-16	1.16E-04	1.02E-02	4.46E-02	8.05E-02	1.22E-02 [20,000 y]	3.57E-01 [20,000 y]	3.93E-01 [20,000 y]
42disp	2X increase in dispersion	3.76E-05	1.81E-16	1.37E-04	1.02E-02	5.17E-02	8.73E-02	1.22E-02 [20,000 y]	3.50E-01 [20,000 y]	3.86E-01 [20,000 y]
Groundwater Cases										
42base[4]	Well intercept. = 1000m	1.58E-05	7.58E-17	5.78E-05	7.58E-03	2.52E-02	5.20E-02	9.29E-03 [76,500 y]	4.03E-01 [100,000 y]	4.37E-01 [100,000 y]
42base[4]	Well intercept. = CR	1.99E-06	9.52E-18	7.26E-06	9.52E-04	3.16E-03	6.53E-03	1.17E-03 [76,500 y]	5.06E-02 [100,000 y]	5.49E-02 [100,000 y]
42base[4]	Trench at south end	3.20E-05	1.53E-16	1.17E-04	1.53E-02	5.09E-02	1.05E-01	1.88E-02 [76,500 y]	8.13E-01 [100,000 y]	8.82E-01 [100,000 y]
42base[4]	90° rotation	4.07E-05	1.95E-16	1.49E-04	1.95E-02	6.47E-02	1.34E-01	2.39E-02 [76,500 y]	1.04E+00 [100,000 y]	1.12E+00 [100,000 y]
42base[4]	Smaller layout	5.75E-06	2.75E-17	2.10E-05	2.75E-03	9.15E-03	1.89E-02	3.38E-03 [76,500 y]	1.46E-01 [100,000 y]	1.59E-01 [100,000 y]
42base[4]	Larger layout	2.56E-05	1.22E-16	9.34E-05	1.22E-02	4.07E-02	8.40E-02	1.50E-02 [76,500 y]	6.50E-01 [100,000 y]	7.06E-01 [100,000 y]
42vault[5]	Existing vault site	2.60E-03	3.95E-10	9.50E-03	7.78E-03	2.98E-01	3.17E-01	7.74E-03 [20,000 y]	3.19E-01 [100,000 y]	3.40E-01 [100,000 y]
42base[4]	30 L/d pumping	2.13E-05	1.02E-16	7.78E-05	1.02E-02	3.39E-02	7.00E-02	1.25E-02 [76,500 y]	5.42E-01 [100,000 y]	5.88E-01 [100,000 y]
42base[4]	100 L/d pumping	2.13E-05	1.02E-16	7.78E-05	1.02E-02	3.39E-02	7.00E-02	1.25E-02 [76,500 y]	5.42E-01 [100,000 y]	5.88E-01 [100,000 y]
42base[4]	300 L/d pumping	2.13E-05	1.02E-16	7.78E-05	1.02E-02	3.39E-02	7.00E-02	1.25E-02 [76,500 y]	5.42E-01 [100,000 y]	5.88E-01 [100,000 y]

Table D-1. Summary of Estimated Impacts for 2001 ILAW PA.

INTEG Filename	Case Designation	Estimated Impact at 1,000 y after Facility Closure (1)			Estimated Impact at10,000 y after Facility Closure (1)			Maximum Estimated Impact (1) [peak time or max time for calculation]		
		B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)
42base(4)	1000 L/d pumping	2.13E-05	1.02E-16	7.78E-05	1.02E-02	3.39E-02	7.00E-02	1.25E-02 [76,500 y]	5.42E-01 [100,000 y]	5.88E-01 [100,000 y]
42base(4)	Reduced hyd. Cond (3X)	2.55E-04	1.22E-15	9.32E-04	1.22E-01	4.06E-01	8.39E-01	1.50E-01 [76,500 y]	6.49E+00 [100,000 y]	7.04E+00 [100,000 y]
42base(4)	Regional recharge increase (3X)	1.96E-05	9.38E-17	7.16E-05	9.38E-03	3.12E-02	6.44E-02	1.15E-02 [76,500 y]	4.99E-01 [100,000 y]	5.41E-01 [100,000 y]
42base(4)	Regional recharge decrease (3X)	1.45E-05	6.94E-17	5.29E-05	6.94E-03	2.31E-02	4.76E-02	8.50E-03 [76,500 y]	3.69E-01 [100,000 y]	4.00E-01 [100,000 y]
42base(4)	Decrease regional upgradient boundaries (2X)	1.79E-05	8.57E-17	6.54E-05	8.57E-03	2.85E-02	5.88E-02	1.05E-02 [76,500 y]	4.55E-01 [100,000 y]	4.94E-01 [100,000 y]
Inventory Cases										
42maxTc	Increase Tc inventory 5X	7.03E-05	1.02E-16	2.98E-04	3.35E-02	3.39E-02	1.75E-01	3.99E-02 [20,000 y]	5.37E-01 [100,000 y]	6.95E-01 [100,000 y]
42noTc	Reduce Tc to 0	9.00E-06	1.02E-16	2.26E-05	4.33E-03	3.39E-02	4.38E-02	6.05E-03 [100,000 y]	5.37E-01 [100,000 y]	5.59E-01 [100,000 y]
42-2i	Double I inventory	3.01E-05	1.02E-16	1.00E-04	1.45E-02	3.39E-02	8.10E-02	1.82E-02 [80,000 y]	5.37E-01 [100,000 y]	6.01E-01 [100,000 y]
42-2u	Double U inventory	2.13E-05	1.02E-16	7.78E-05	1.02E-02	5.82E-02	7.55E-02	1.24E-02 [52,000 y]	8.72E-01 [96,000 y]	6.77E-01 [100,000 y]
42ub	All inventory at bounding	5.52E-05	4.83E-16	1.73E-04	2.66E-02	1.86E-01	3.11E-01	3.47E-02 [100,000 y]	2.76E+00 [100,000 y]	2.95E+00 [100,000 y]
42ubmaxTc	Bounding value + max Tc	1.02E-04	4.83E-16	3.85E-04	4.90E-02	1.86E-01	4.12E-01	5.96E-02 [50,000 y]	2.76E+00 [100,000 y]	3.07E+00 [32,000 y]

Table D-1. Summary of Estimated Impacts for 2001 ILAW PA.

INTEG Filename	Case Designation	Estimated Impact at 1,000 y after Facility Closure (1)			Estimated Impact at10,000 y after Facility Closure (1)			Maximum Estimated Impact (1) [peak time or max time for calculation]		
		B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)
Dosimetry Cases										
ind-100m	Industrial	9.86E-06	1.02E-16	9.86E-06	4.71E-03	3.39E-02	1.69E-02	5.71E-03 [53,000 y]	5.37E-01 [100,000 y]	2.08E-01 [100,000 y]
res-100m	Residential	2.88E-05	1.02E-16	3.44E-05	1.38E-02	3.39E-02	5.68E-02	1.68E-02 [53,000 y]	5.37E-01 [100,000 y]	6.89E-01 [100,000 y]
agr-100m	Agricultural	2.88E-05	1.02E-16	1.26E-04	1.38E-02	3.39E-02	1.01E-01	1.68E-02 [53,000 y]	5.37E-01 [100,000 y]	7.48E-01 [100,000 y]
nat-CR	Native American (3)	4.52E-06	1.07E-17	2.77E-05	2.16E-03	3.55E-03	2.22E-02	2.63E-03 [52,500 y]	5.62E-02 [100,000 y]	1.64E-01 [100,000 y]
CRPop-CR	CR population (3)	1.12E-02	1.07E-17	3.42E-02	5.33E+00	3.55E-03	3.28E+02	6.49E+0 [53,000 y]	5.62E-02 [100,000y]	2.91E+02 [100,000 y]
DOE-100m	DOE dose parameters	1.87E-05	9.55E-17	6.67E-05	8.96E-03	3.16E-02	5.93E-02	1.10E-02 [58,000 y]	5.01E-01 [100,000 y]	4.91E-01 [100,000 y]
Other Sensitivity Cases										
42sub	Cap break subsidence	5.33E-09	2.56E-20	1.95E-08	3.97E-03	2.35E-05	1.45E-02	5.22E-03 [19,000 y]	1.28E-02 [20,000 y]	3.15E-02 [20,000 y]
Pulse42	Pulse	7.41E+00	3.56E-11	2.71E+01	5.18E-12	6.50E+01	6.29E+01	1.69E+01 [1,385 y]	6.93E+01 [11,100 y]	6.72E+01 [11,200 y]
42instant (2)	Instantaneous VZ transport	2.87E-03	1.48E+0 (3.49E-04)	3.39E+00	1.61E-02	4.68E+00 (6.4E-03)	9.36E+00	7.96E-01 [33 y]	4.83E+00 [13,100 y] (5.16E-02 [100,000 y])	9.43E+00 [11,100 y]
42conc	Concrete K_ds	1.37E-05	1.02E-16	5.87E-05	9.98E-03	5.56E-04	3.73E-02	1.24E-02 [53,000 y]	4.74E-01 [100,000 y]	5.22E-01 [100,000 y]

Table D-1. Summary of Estimated Impacts for 2001 ILAW PA.

INTEG Filename	Case Designation	Estimated Impact at 1,000 y after Facility Closure (1)			Estimated Impact at10,000 y after Facility Closure (1)			Maximum Estimated Impact (1) [peak time or max time for calculation]		
		B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)	B/γ dose (mrem/y)	alpha concen. (pCi/L)	all-pathways dose (mrem/y)

(1) The estimated impacts are associated with a well located 100 meters downgradient form the ILAW site

(2) The numbers in parentheses are the estimated radium concentrations

(3) The estimated impacts are calculated for a well just before the aquifer flows into the Columbia River

(4) The estimated impacts use the WIFs provided in Tables 4-2 and 4-16 with the Base Analysis Case

(5) The estimated impact use the existing vault WIF provided in Table 4-16 with the Base Analysis Case WIF from Table 4-2

D.6 Outline for Compact Disk Files

The output files for each sensitivity case have been copied onto a compact disk (CD) that is available on request. Table D-2 provides a cross-reference between the sensitivity case descriptions and the summary output files for the waste form, vadose zone, and dose estimate (INTEG) results.

All files on the CD are stored under a directory structure that defines the type of files and case names. The top directory is named 'ILAW;' beneath it are two directories named 'Waste Form' and 'Integration.' The 'Waste Form' directory contains subdirectories corresponding to different waste form cases, which in turn contain the data files used in the calculations. The 'Integration' directory contains subdirectories corresponding to the INTEG cases, which in turn contain the input and output files for each INTEG calculation.

The waste form case subdirectory names begin with the name of the STORM case (e.g., case WFA is in directory WFA-base-42) identified in Table D-2. Each subdirectory contains two files: 'storm2vam.out' and 'storm2vam.g17.' Some subdirectories may contain multiple versions of each file, with a suffix indicating which near-field flow calculation they are associated with. The '.out' files contain excerpted data from the STORM output data. These data are in three columns, corresponding to time (year), release rate (micromols/m^2/secomd), and release rate (Ci/m^2/year). The '.g17' files contain the release rate data (converted to Ci/year for each node) in the format required for input to the far-field transport calculations. The integration subdirectory names correspond to the INTEG cases identified in Tables D-1 and D-2. Each subdirectory contains four files: named 'run.bat,' 'vad casename.bac,' 'casename.out,' and 'casename.dat' (where 'casename' is replaced with the case name). The 'run.bat' file contains some of the input parameters required for the INTEG run, including the well intercept factor (WIF), recharge rate, and the names of the data files used in the INTEG calculation (all the data files are in the 'lib' subdirectory under the integration directory). The 'vad-casename.bac' file contains the data from the corresponding far-field transport calculations. These data are the release rate (in Ci/year) from the vadose zone into the groundwater with the first column being time (in years) and the subsequent columns each corresponding to a different K_d (typically only the first two K_d columns are used, with the remainder being filled with 0s). The 'casename.out' file contains the general results of the INTEG calculation and the 'casename.dat' file contains the detailed results for each nuclide.

Table D-2. Cross-reference for Sensitivity Cases and Output Files from Waste Form, Vadose Zone, and INTEG Calculations for the 2001 ILAW PA.

Case	Description	Waste Form Filename	Far Field Transport Filename	INTEG Case Filename
1	**Reference Cases:**			
1-1	Base analysis case	wfa	42base-t	42base
1-2	Best estimate case	wfd	42break-t	42break
2	**Scenario Cases:**			
2-1	Irrigation	wfa	42base-t	50 [(a)]
2-2	Well locations	wfa	42base-t	42base [(a)]
2-3	Pumping rates	wfa	42base-t	42base [(a)]
3	**Inventory Cases:**			
3-1	Upper bound inventory.	wfa	42base-t	42ub
3-2	Maximum ^{99}Tc inventory	wfa	42base-t	42maxtc
3-3	No Tc inventory	wfa	42base-t	42notc
3-4	Double I inventory	wfa	42base-t	42-2I
3-5	Double U inventory	wfa	42base-t	42-2u
3-6	Upper bound + maximum Tc inventory	wfa	42base-t	42ubmaxtc
4	**Recharge Cases:**			
4-1	Rupert sand	wf4	09-t	09
4-2	High recharge	wf6	50-t	50
4-3	Low recharge	wfd	01-t	01
4-4	Low recharge for 500 years, then 4.2mm/year	wfa [(b)]	42base-t [(b)]	42base [(b)]
5	**Geology Cases:**			
5-1	Entire formation - sandy	wfa	42sand-t	42sand
5-2	Entire formation - gravelly	wfa	42grav-t	42grav
5-3	Clastic dikes	wfa	42dike-t	42dike
5-4	Deep water table	wfa	42deep-t	42deep
5-5	Instantaneous vadose zone transport	wfa	na [(c)]	42instant
6	**Facility Cases:**			
6-1	Capillary break (best estimate case)	Same as case 1-2		
6-2	Side slope	wfa	42all-t	42all
6-3	Vertical capillary break	wfa	42vert-t	42vert
6-4	Short break	wfa	42short-t	42short
6-5	Sand backfill	wf11	42sandfill-t	42sandfill
6-6	Alternative facility design (vault)	wf16	42vault-t	42vault

Table D-2. Cross-reference for Sensitivity Cases and Output Files from Waste Form, Vadose Zone, and INTEG Calculations for the 2001 ILAW PA.

Case	Description	Waste Form Filename	Far Field Transport Filename	INTEG Case Filename
7	**Degradation Cases:**			
7-1	Surface barrier	no barrier = base case		
7-2	Degraded capillary break (subsidence)	wfa		42sub
7-3	Bathtub effect	(d)	(d)	(d)
8	**Hydrologic Parameter Cases:**			
8-1	Isotropic conductivity	wfa	42iso-t	42iso
8-2	Sand backfill	Same as case 6-5		
8-3	Entire formation - sandy	Same as case 5-1		
8-4	Entire formation - gravelly	Same as case 5-2		
9	**Waste Form Cases:**			
9-1	Forward rate	wfb	42for-t	42for
9-2	No ion exchange	wf1	42noion-t	42noion
9-3	No secondary product	wf2	42nosec-t	42nosec
9-4	Infiltration rate = 0.1 mm/y	Same as case 4-3		
9-4	Infiltration rate = 0.5 mm/y	wf3	nc [e]	nc [e]
9-5	Infiltration rate = 0.9 mm/y	Same as case 4-1		
9-6	Infiltration rate = 4.2 mm/y	Same as case 1-1		
9-7	Infiltration rate = 10 mm/y	wf5	nc [e]	nc [e]
9-8	Infiltration rate = 50 mm/y	Same as case 4-2		
9-9	Sand backfill	Same as case 6-5		
9-10	Steel containers	wf25	42steel-t	42steel
9-11	Chemical conditioning layer	wf10	42cond-t	42condlayr
9-12	Alternative facility (vault)	Same as case 6-6		
9-13	Alternative glass	wf28	42altglass-t	42altglass
9-14	Diffusion parameter	wf29	42sdiff-t	42sdiff
9-15	Uranium	wf26	nc [e]	nc [e]
9-16	Two-dimensional waste form model	wf9	nc [e]	nc [e]
9-17	Pulse	na [c]	pulse-t	pulse42
10	**Geochemical Cases:**			
10-1	Uranium trapping	wfa	42base-t, 42conc-t	42conc
10-2	Uranium k_d = 0.0 mL/g	wfa	42base-t	42ukd0
10-3	All k_d = 0.0 mL/g	wfa	42base-t	42kd0
11	**Exposure**			
11-1	Alternative dose factors (DOE)	wfa	42base-t	DOE-100m
11-2	Industrial scenario - 100 m	wfa	42base-t	ind-100m

Table D-2. Cross-reference for Sensitivity Cases and Output Files from Waste Form, Vadose Zone, and INTEG Calculations for the 2001 ILAW PA.

Case	Description	Waste Form Filename	Far Field Transport Filename	INTEG Case Filename
11-3	Residential scenario - 100 m	wfa	42base-t	res-100m
11-4	Agricultural scenario - 100 m	wfa	42base-t	agr-100m
11-5	Native American - CR	wfa	42base-t	nat-CR
11-6	CR population - CR	wfa	42base-t	CRPop-CR
12	**Location/Layout of Facility**			
12-1	Trench layouts	wfa	42base-t	42base [(a)]
12-2	Existing vaults	wf16	42vault-t	42vault [(f)]

(a) Use results from base analysis case and normalize INTEG results to appropriate WIFs provided in Table 4-2 and 4-16

(b) Estimated from base analysis case @ 500 years earlier (e.g., at 500 and 9,500 years after facility closure)

(c) na = not applicable

(d) See Bacon 2001 and Finfrock 2000b for details

(e) nc = not calculated

(f) Use results from vault calculation and WIF for existing vault site (Table 4-16)

Appendix E

A Strategy to Conduct an Analysis of the Long-Term Performance of Low-Activity Waste Glass in a Shallow Subsurface Disposal System at Hanford

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SUMMARY

Treatment services are being procured to vitrify low-activity tank waste for eventual disposal in a shallow subsurface facility at the Hanford Site. Over 500,000 metric tons of low-activity waste glass will be generated, which is among the largest volumes of waste within the U.S. Department of Energy (DOE) complex and is one of the largest inventories of long-lived radionuclides planned for disposal in a low-level waste facility. Before immobilized waste can be disposed of, DOE must approve a "performance assessment," which is a document that describes the impacts of the disposal facility on public health and environmental resources. Because the release rate of radionuclides from the glass waste form is a key factor determining these impacts, a sound scientific basis for determining their long-term release rates must be developed if this disposal action is to be accepted by regulatory agencies, stakeholders, and the public. In part, the scientific basis is determined from a sound testing strategy.

The foundation of the proposed testing strategy is a well-accepted mechanistic model that is being used to calculate the glass corrosion behavior over the geologic time scales required for performance assessment. This model requires that six parameters be determined, and the testing program is defined by an appropriate set of laboratory experiments to determine these parameters, and is combined with a set of field experiments to validate the model as a whole.

Three general classes of laboratory tests are proposed in this strategy: 1) characterization, 2) accelerated, and 3) service condition. Characterization tests isolate and provide specific information about processes or parameters in theoretical models. Accelerated tests investigate corrosion behavior that will be important over the regulated service life of a disposal system within a laboratory time frame of a few years or less. Service condition tests verify that the techniques used in accelerated tests do not change the alteration mechanisms. The recommended characterization tests are single-pass flow-through tests using a batch reactor design. Accelerated and service conditions tests include product consistency and pressurized unsaturated flow (PUF) tests. Nonradioactive glasses will be used for the majority of the laboratory testing (~80%), with the remainder performed with glasses containing a selected set of key radionuclides. Additionally, a series of PUF experiments with a natural analog of basaltic glass is recommended to confirm that the alteration products observed under accelerated conditions in the PUF tests are similar to those found associated with the natural analog. This will provide additional confidence in using the PUF test results to infer long-term corrosion behavior.

Field tests are proposed as a unique way to validate the glass corrosion and contaminant transport models being used in the performance assessment. To better control the test conditions, the field tests are to be performed in lysimeters (corrugated steel containers buried flush with the ground surface). Lysimeters provide a way to combine a glass, Hanford formation soil, and perhaps other engineered materials in a well-controlled test, but on a scale that is not practicable in the laboratory. The recommended field tests include some experiments where a steady flow rate of water is artificially applied. These tests use a glass designed to have a high corrosion rate, making monitoring contaminant release and transport easier. Either existing lysimeters at the Hanford Site or new lysimeters that have been equipped with the latest in monitoring equipment and located near the proposed disposal site can be used for these experiments.

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GLOSSARY

ADT	accelerated dissolution test
ALT	accelerated leach test
ANL	Argonne National Laboratory
AREST-CT	Analyzer of RadionuclidE Source Term with Chemical Transport
ASTM	American Society for Testing and Materials
CPU	central processing unit
DLT	dynamic leach test
DOE	U.S. Department of Energy
EMSP	Environmental Management Science Program
GFLOPS	giga floating point operations per second
HLW	high-level waste
ILAW	immobilized low-activity waste
ISO	International Standards Organization
LAW	low-activity waste
MIIT	Materials Interface Interaction Test
PCT	product consistency test
PNNL	Pacific Northwest National Laboratory
PUF	pressurized unsaturated flow
RH	relative humidity
SPFT	single-pass flow-through test
STOMP	a nonisothermal, multiphase flow simulator
STORM	Subsurface Transport Over Reactive Multiphases
S/V	surface area-to-solution volume ratio
VHT	vapor hydration test
WIPP	Waste Isolation Pilot Plant

E-1.0 INTRODUCTION

The Hanford Site in southeastern Washington State has been used extensively to produce nuclear materials for the U.S. strategic defense arsenal by the U.S. Department of Energy (DOE) and its predecessors, the U.S. Atomic Energy Commission and the U.S. Energy Research and Development Administration. A large inventory of radioactive and mixed waste has accumulated in 177 buried single- and double-shell tanks. The DOE is proceeding with plans to permanently dispose of this waste (Ecology 1996; 62 FR 8693). Liquid waste recovered from the tanks will be pretreated to separate the low-activity fraction from the high-level and transuranic waste. The small volume of high-level immobilized waste and the much larger volume of low-activity waste (LAW) will be disposed of in different locations. The high-level waste (HLW) will be stored on the Hanford Site until sent to a federal geologic repository. The immobilized low-activity waste (ILAW) will be placed in a near-surface disposal system on the Hanford Site.

Services are being procured for treating and immobilizing the tank waste. The leading processing option for waste immobilization is vitrification. Vitrifying the LAW is expected to generate over 500,000 metric tons or 200,000 m^3 (6,000,000 ft^3) of ILAW glass that will be disposed of under this plan. This is among the largest volumes of waste within the DOE complex and is one of the largest inventories of long-lived radionuclides planned for disposal in a low-level waste facility.

Before the immobilized waste can be disposed of, DOE must approve a "performance assessment," which is a document that describes the long-term impacts of the disposal facility on public health and environmental resources. The first ILAW performance assessment was published in 1998 (Mann 1998a), and DOE conditionally approved this performance assessment (DOE 1999d), issuing a disposal authorization statement (DOE 1999a). A major conclusion from the performance assessment is that the release rate of radionuclides from the glass waste form by reaction with water is one of the key parameters that determines the impacts of the disposal action and is the most uncertain. Consequently, a sound scientific basis for determining the long-term release rates of radionuclides from these glasses must be developed if this disposal action is to be accepted by regulatory agencies, stakeholders, Native American Tribes, and the public.

A general approach for the evaluation of materials behavior in a disposal site has been developed that outlines logical steps to validate and confirm the corrosion behavior of materials whose life expectancies must greatly exceed the length of time over which experimental data can be obtained (ASTM 1991). These steps include determining the likely range of environmental factors in the disposal system, identifying and characterizing materials that are likely to be present in the disposal system, performing tests under site-relevant conditions to determine important alteration processes for those materials, developing models for key alteration processes, and performing tests that accelerate those processes. The ASTM protocol also recommends tests to confirm the corrosion model and to use information provided by analog materials or systems. Many steps in this approach relevant to waste glass behavior in general

have been completed in studies conducted for deep geologic disposal systems and can be directly applied to the Hanford Site disposal system. For example, the processes that control glass corrosion are well understood, and rate expressions have been developed and tested. What remains is primarily to characterize the specific corrosion behavior of actual ILAW glasses. This includes identifying (or confirming) the corrosion processes that will control the long-term behavior of the glasses and release of radionuclides, measuring parameters needed to perform model calculations for performance assessment, and conducting accelerated and service condition tests to confirm and provide confidence in those calculations.

The purpose of this report is to document a technical strategy developed by Pacific Northwest National Laboratory (PNNL)[a] and Argonne National Laboratory that the ILAW disposal program will pursue over the next several years to evaluate the long-term radionuclide release behavior of the ILAW glass(es) under development by the private contractors. As such, this document is intended to serve as the technical basis for the glass testing program that is needed to complete the performance assessment for the ILAW disposal facility. Specific matrices of tests will not be developed in this report; these are to be provided in separate test plans for each test method. However, this document will define the general classes of tests for which test plans will need to be developed and provide a general range of appropriate conditions for these tests. We begin the discussion of the technical strategy with a brief overview of the disposal system design and expected environmental conditions at the site.

[a]Pacific Northwest National Laboratory is operated for the U.S. Department of Energy by Battelle under Contract DE-AC06-76RLO 1830.

E-2.0 DISPOSAL SYSTEM DESCRIPTION

The Hanford Site is a 1450 km^2 area of semi-arid land located in southeastern Washington state. Average annual precipitation is 16 cm, with 44% of this total occurring during November, December, and January. Daytime high temperatures in summer can exceed 40°C, while outbreaks of arctic air masses in winter can cause temperatures to drop below -18°C. Plans call for the disposal system to include a protective surface barrier with design elements to minimize root intrusion, animal intrusion, and water infiltration. The use of silt-loam soils, when combined with a representative community of shrub-steppe vegetation, has been shown to cause most precipitation falling on the region to be lost through evapotranspiration. Consequently, the disposal facility is to be located in relatively dry, unsaturated soil, and performance assessment models must be applicable to the specific physics and chemistry of this type of system.

E-2.1 WATER INFILTRATION

Water flow in the near-surface unsaturated zone is transient because of intermittent precipitation events. Transient water flow begins when water enters at the ground surface and infiltrates downward into the soil column. At some distance from the ground surface transient effects will dampen out and the downward flowing water will reach a steady infiltration rate. The distance at which steady infiltration occurs is sometimes referred to as the penetration depth (Eagleson 1978; Salvucci 1993). Thus, the unsaturated zone essentially comprises two regions: an unsteady-flow region between the ground surface and penetration depth, and a steady-flow region between the penetration depth and the saturated zone water table. The steady flux in the lower unsaturated region is equal to the annual rate of groundwater recharge and therefore is composed of contributions not only from the most recent pulse, but from previous precipitation events as well.

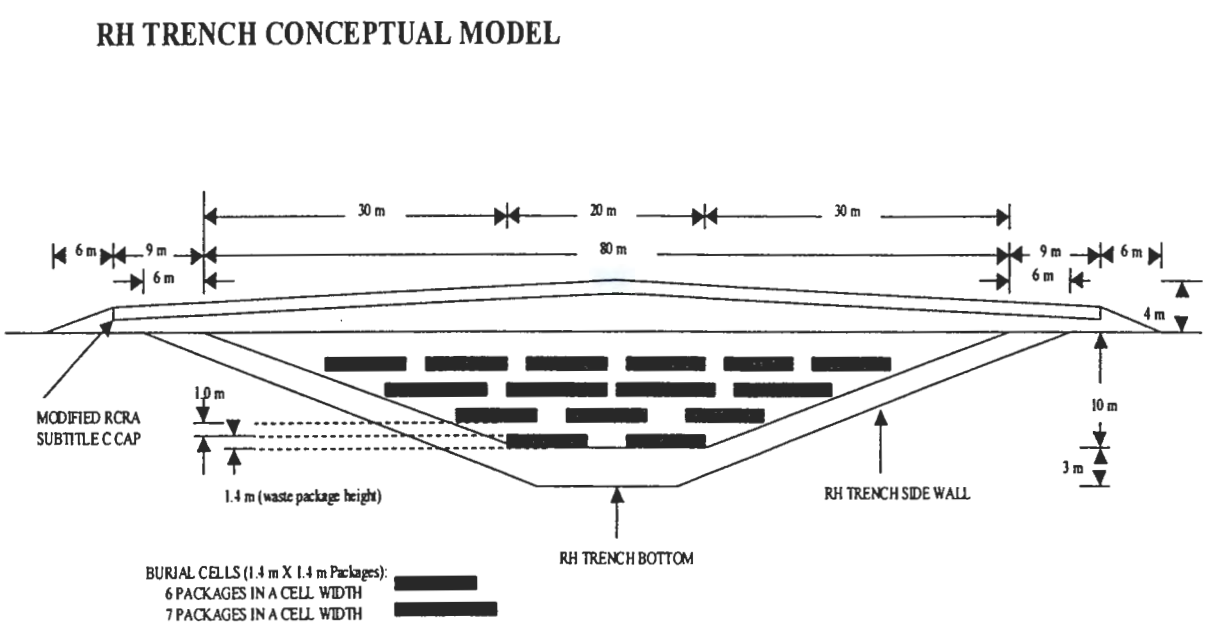
The ILAW disposal facility will be situated below the penetration depth in the region of steady flow. The natural rate of moisture infiltration is approximately 4.2 mm/year (Fayer 1999). However, the natural rate of moisture infiltration cannot be relied upon because construction of the disposal system will destroy the natural soil-sediment profile and remove surface vegetation. Consequently, a protective surface barrier will be engineered with sediment layers and a capillary barrier to prevent or minimize infiltration. The design basis for the Hanford barrier (Myers and Duranceau 1994) specifies that less than 0.5 mm/year will pass through the barrier for the first 1,000 years. Infiltration beyond the root zone is controlled at the soil/atmosphere interface where surface soils and sediments and vegetation interact with the climate. The frequency, duration, and magnitude of precipitation and runoff events determine the infiltration rate. Infiltration into the disposal facility will be controlled by the physics described above, the unsaturated hydraulic properties of the surface and subsurface barriers to infiltration, the vault and surrounding soil, and the matric and gravity potential gradients.

E-2.2 FACILITY DESCRIPTION

A detailed design for the ILAW disposal facility is not yet available. However, the current designs (Puigh 1998) have the disposal facility as a series of large, covered trenches. The present plans are that the ILAW packages will be 1.22 m in diameter by 2.3 m high. A layer of ILAW packages will be laid down and covered with dirt. Across the center of the facility, packages would be arranged in three layers, with only one or two layers along the outer edges. Figure E-2.1 shows a schematic for the disposal facility, using a slightly earlier design.

All the concepts include backfilled soil around and on top of the waste containers in the facility. The soil was included in these concepts 1) for structural support, 2) to wick moisture away from the waste containers, and 3) to provide radiation shielding for the facility workers. These concepts also have a similar barrier philosophy. The uppermost barrier is the surface barrier, which consists of sand-gravel layers to work as a capillary break, a layer (that may or may not be included in the final design) for conditioning the chemistry of the water to help minimize the rate of glass corrosion and/or radionuclide release, and a layer of basalt riprap to deter burrowing animals, plant root intrusion, and inadvertent intruders. Beneath the surface barrier, another sand-gravel capillary break will divert moisture coming through the surface barrier away from the vault. This barrier combination minimizes the amount of water that enters the vault.

Figure E-2.1. Schematic of ILAW Disposal System. (This is for the design from early 2000).



The disposal vaults are to be located approximately 15 m below the top of the surface barrier. At this depth, the ambient temperature is approximately 15°C, and temperature fluctuations are less than 2°C. Also, because the ILAW generates a small amount of heat from radioactive decay (McGrail and Mahoney 1995), the disposal system can be treated adequately as an isothermal system. The waste packages are expected to consist of a rectangular steel container that holds the glass and possibly a filler material. The glass will likely not be annealed and thus will be thermally stress fractured to some extent. The extent of fracturing is important because it increases the available surface area for possible contact by water and thus can potentially increase the radionuclide release rate from the disposal system.

Based on this disposal system concept, small amounts of water are expected to percolate through the disposal system and eventually penetrate the steel containers and contact the glass. The goal of the glass testing program then, is to provide the necessary supporting data so that the radionuclide release behavior of this system can be modeled with sufficient confidence to support an overall performance assessment for disposal of ILAW at the Hanford Site.

E-3.0 MODELING APPROACH

Because it is impossible to formulate a rationale and a defensible testing program without understanding the physical and chemical processes that govern glass corrosion and radionuclide release as well as the conceptual models that describe these processes, we briefly review these concepts below.

E-3.1 GLASS CORROSION PROCESS

Fortunately, much is known about glass-water reaction processes. Glass forms ranging from simple binary and ternary silicate glasses to complex waste glasses with 30 or more components have been studied for 35 years. Much of this work relevant to silicate waste glasses has recently been compiled and critically reviewed (Bates et al. 1994). A general picture of the glass corrosion process in water has emerged that can be summarized as follows: on initial contact by water, alkali is extracted by ion exchange in what is thought to be a diffusion-controlled process. Simultaneously, hydrolysis and dissolution of the glass network occurs. In unsaturated disposal systems where water content and flow rate are expected to be very low, the rate of ion exchange and dissolution decrease, but for different reasons. The ion-exchange rate slows in accordance with a diffusion-controlled process as a reaction layer builds up on the glass over time. A reaction layer builds up as a result of silanol condensation reactions that reform Si-O bonds. The dissolution rate of the glass network slows because of the common ion effect (i.e., as the solution becomes more concentrated in glass components, the difference in chemical potential between the glass and aqueous phase decreases, which decreases the dissolution rate). The dissolution rate cannot become zero because silicate glasses are thermodynamically unstable in water.

E-3.1.1 Importance of Secondary Phases

As a solution in contact with a dissolving glass becomes more and more concentrated in glass components, solubility limits for alteration phases begin to be exceeded. Although no generally accepted theory has been proposed to describe the factors controlling nucleation and growth, ultimately, the glass transforms into a paragenetic assemblage of alteration products or minerals. Also, no generally accepted theory exists that can be used to predict which specific phases should form or the sequence of their formation. However, from the Simplexity Principle (Goldsmith 1953), we know that the most disordered phase should form from a random system of components first. We know from the Ostwald Step Rule (Ostwald 1897) that subsequent transformations to more stable phases should occur in steps where the reaction products obtained at each step lay near the previous state in free energy. These governing principles have extremely important implications for the testing strategy because at the low temperatures (15°C) relevant for the ILAW disposal system, metastable and/or amorphous phases may persist for long periods of time. Consequently, a means must be found for accelerating the transformation process during laboratory testing without altering the fundamental transformation process itself.

The secondary phases that form from the glass-water reaction process are expected to depend principally on the composition of the glass and not on other components in the disposal system because the glass supplies the majority of the elements to the fluid from which the secondary phases precipitate. However, the water conditioning layer, filler material in and/or between waste packages, and the concrete used in vault construction also may affect the formation of secondary phases, especially in localized regions. Glasses that are stable with respect to the formation of alteration products will maintain a slow but finite rate of network hydrolysis and dissolution indefinitely. Many existing natural glasses exhibit these characteristics, having withstood weathering over geologic time scales. Laboratory tests have generally reproduced the same types of alteration products that have been found on these glasses (Grambow et al. 1986, Luo et al. 1997), confirming their long-term stability with respect to forming alteration phases. However, glasses that are unstable with respect to alteration product formation exhibit autocatalytic reactivity, i.e., a very rapid increase in dissolution rate that is limited only by the availability of water or the forward reaction rate of the glass, whichever is the rate limiting process. High-level waste glasses (Van Iseghem and Grambow 1988 have exhibited this phenomenon and so has a representative ILAW glass, LD6-5412 (McGrail et al. 1997a). Consequently, the laboratory testing program must ensure that the ILAW glass(es) being produced by the private vendors fall into the former, stable category. If not, then the tests should provide guidance for modifying the glass composition into a region with known long-term stability.

E-3.1.2 Effect of Ion Exchange

Because of the large effect that secondary phase formation has on glass dissolution rates, recent work on glass-water interactions has focused on understanding this process and incorporating it into models (Ebert and Bates 1993). The ion-exchange process has been largely ignored because it has been thought to be a short-duration, secondary, or tertiary process that had little or no bearing on long-term corrosion or radionuclide release rates from glasses. The only significant effect identified in the literature that is attributed to alkali ion exchange is an increase in solution pH in static laboratory tests conducted at high surface-area-to-volume ratios (Strachan et al. 1990; Bourcier and Feng 1993).

The discovery of the significance of ion exchange to long-term radionuclide release rates was only very recently made possible by the development of computational tools that can simulate the coupled processes of glass dissolution, mass transport, and chemical reactions in a complex disposal system (Chen, McGrail, and Engel 1997). By comparing simulations where Na ion exchange was included versus those where it was excluded, Chen, McGrail, and Engel (1997) showed that the ion-exchange reaction increased the pH in a conceptual design for an ILAW disposal vault from approximately 9.8 to over 12.5, which increased the radionuclide release rate by over 3 orders of magnitude. Thus, the overall performance of the disposal system can be significantly improved if improved glasses are developed that minimize alkali ion exchange. Formulation of such glasses can be rationally accomplished only through development of a fundamental scientific understanding of the ion-exchange process. Such a study is currently

being supported under DOE's Environmental Management Science Program (EMSP), and the results and conclusions from this work will be closely coordinated with the testing program described in this report.

E-3.2 GLASS CORROSION MODEL

It is impossible to develop a rationale testing program without thoroughly understanding the model that is being used to calculate the glass corrosion behavior over the geologic time scales required for performance assessment. In fact, the testing program is directly linked and derived from the data requirements of this model. Figure E-3.1 illustrates the basic link between modeling and testing. At the top of the pyramid is the conceptual model that describes the glass corrosion process. The parameters that make up this model are abstracted at the mid-level of the pyramid. Once the parameter set is known, the testing program is determined by the appropriate experiments needed to obtain each parameter in the set. This strategy is consistent with American Society for Testing and Materials guidelines (ASTM 1991), but we emphasize the "model driven" approach here because, ultimately, the credibility of the performance assessment rests on the credibility of the models and supporting data, not on a specific test. Following this approach, the conceptual model for glass corrosion planned for use in the performance assessment is discussed in the next section.

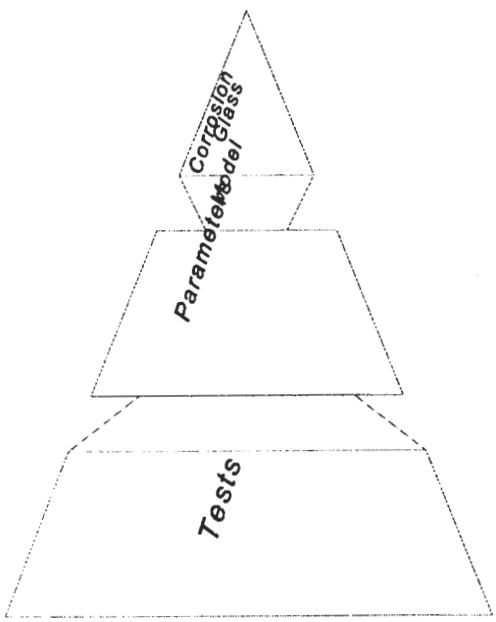


Figure E-3.1. Links Among Glass Tests and Modeling.

E-3.2.1 Rate Law for Hydrolysis and Dissolution

The literature on modeling glass dissolution is extensive. However, Strachan, Bourcier, and McGrail (Strachan et al.1994) published a review of the subject. An interested reader should consult this paper and the references cited therein for additional details. The conclusion from this study was that of all the models that have been developed to describe glass dissolution behavior, the general kinetic rate law proposed by Aagaard and Helgeson (Aagaard and Helgeson 1982) and later adapted by Grambow (Grambow 1985), best describes the majority of the experimental data that has been gathered over 35 years of studying glass-water reaction processes. Consequently, this model has been selected for use on the ILAW disposal project. The corrosion of silicate glasses in water can be represented as a special type of irreversible dissolution reaction. The reaction is irreversible because the glass cannot be reformed by precipitation from aqueous solution.

A conventional transition state kinetic rate equation can be used to compute the flux of any element *i* released from the glass into the aqueous phase and is given by

$$J_i^a = v_i k e^{-\frac{E_a}{RT}} \left[1 - \left(\frac{Q}{K} \right)^{\sigma} \right] \prod_{j=1...M} a_j^{-\eta_j} \quad i1,2,...N \tag{E.1}$$

where

a_j	=	activity of *j*th aqueous species
k	=	intrinsic rate constant, $g/m^2 \bullet s$
E_a	=	activation energy, J/mol
J_i^a	=	flux of element *i* to the aqueous phase, $g/m^2 \bullet s$
K	=	equilibrium constant of rate controlling reaction
M	=	number of species directly affecting the rate
N	=	number of elements
Q	=	ion-activity product of rate controlling reaction
R	=	gas constant, 8.314 J/mol•K
T	=	temperature, K
v_i	=	stoichiometric coefficient of element *i* in the glass
σ	=	net reaction order
η_j	=	stoichiometric coefficient for the *j*th reactant species.

Equation (E.1) is a constitutive relationship that relates temperature and the composition of water contacting the glass to the corrosion rate. Currently, H^+ is the only aqueous species that has been found to directly influence the rate via the activity product term (Bourcier et al. 1992). Consequently, Equation (E.1) can be simplified to

$$J_i^a = v_i k e^{-\frac{E_a}{RT}} a_{H^+}^{-\eta} \left[1 - \left(\frac{Q}{K} \right)^{\sigma} \right], \qquad i = 1,2,...N \tag{E.2}$$

where a_{H^+} is the hydrogen ion activity. Because the temperature is assumed to be a known constant, and v_i values are determined from the glass composition, application of Equation (E.2) for modeling glass corrosion in a disposal system requires the determination of six parameters; k, E_a, η, σ, K, and Q. In accordance with the strategy illustrated in Figure E-3.1 and the ASTM 1174 approach, the testing program is then defined by an appropriate set of experiments that determines these parameters.

E-3.2.1.1 Experiments for Kinetic Rate Law Parameters. If a glass is placed in a dilute solution that is refreshed at an infinite rate, the chemical affinity term $[1-(Q/K)^{\sigma}]$ in Equation (G.2) equals 1, and the corrosion rate of the glass is determined only by the three traditional kinetic rate law parameters, k, E_a, and η. Experimentally, the single-pass flow-through (SPFT) test is used to approximate these conditions. By adjusting glass surface area and flow rate, the solution remains dilute, but not so dilute that the solution concentrations cannot be determined with standard analytical techniques. The SPFT test has been used to measure reaction rates of

minerals (Knauss and Wolery 1986; Dove and Crerar 1990) and glasses (Knauss et al. 1990; McGrail and Olson 1992; McGrail et al. 1997c). By monitoring the change in dissolution rate over a sufficient range of temperature and pH values, k, E_a, and η can be easily obtained by applying standard nonlinear regression techniques to Equation (E.2). Details of the recommended experimental procedure and tests are given in Section E-4.1.

E-3.2.1.2 Experiments for Affinity Term Parameters. By definition, σ and K are the *parameters* of the chemical affinity term $[1-(Q/K)^{\sigma}]$ and so are usually obtained from laboratory experiments. The ion activity product (Q) is a *variable* and must be computed as a function of time and space for the disposal system (McGrail and Mahoney 1995; Bacon and McGrail 1997). Computation of Q is complex and depends on physical properties of the system, such as flow rate and glass surface area, and chemical properties, such as solubility products and the amounts and types of alteration products formed (see Section E-3.1.1). Because transport and chemical processes interact, or more precisely are coupled, a special type of computational model, called a reactive chemical transport model, is required for simulations.[a] The key inputs to this model derived from the glass testing program are the parameters σ and K, and the identification of glass alteration products. These data are extremely important because they can affect the calculated long-term corrosion rate of the glass by 3 to 4 orders of magnitude.

The pseudoequilibrium constant (K) and reaction order parameter (σ) probably are the most difficult parameters to obtain in the rate law, and, unfortunately, also are the most poorly documented in terms of experimental techniques and methods for obtaining them. Unlike for a crystalline or amorphous phase, the assignment of K for a glass is ambiguous, because by definition, equilibrium between the glass and water does not exist. Assigning K to a hypothetical phase including all the glass elements has not proven successful in modeling laboratory test data (Bourcier 1989; Advocat et al. 1990). Consequently, an approximation is used where K is associated with a reversible microscopic reaction that is rate limiting and not the macroscopic glass-water reaction itself. Good agreement with laboratory test data for most glasses has been obtained by assigning K to a simple SiO_2 polymorph, such as chalcedony, or mixtures of simple hydroxides and silicate phases (Bourcier et al. 1990; Gin 1996).

Unfortunately, the "agreement" typically reported in the literature lacks a statistical basis and usually is just trial and error reckoning of batch test data to calculated curves (Bourcier et al. 1990). The situation for determining the parameter σ is even poorer with most studies of silicate mineral and glass dissolution simply assuming $\sigma=1$. In other cases (Bourcier et al. 1994), σ and/or K have been regressed using data from batch tests. Batch tests introduce additional difficulties in accurately obtaining these parameters because pH and solution concentrations change continuously throughout the test, requiring "corrections" derived from the rate law itself. This introduces the possibility of inseparable causality errors. In a recent paper, McGrail et al. (1997c) illustrated how the SPFT test can be used to obtain σ and K by varying the flow rate over a sufficiently wide range. Their technique has the advantage of being performed at fixed pH and steady-state solution concentrations and does not require data "corrections" derived from the rate

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[a]The interested reader should consult McGrail and Mahoney (1995), Chen et al. (1997), and Bacon and McGrail (1997) for details.

law. Consequently, the experimental techniques and formal mathematics developed by McGrail et al. will be used to obtain these parameters (see Section E-4.1).

Because there is no single test that can reliably evaluate the effects of secondary phases on the long-term corrosion rate of a glass, our approach is to employ a range of accelerated experimental techniques. By doing so, a range of physical and environmental factors can be examined that affect the rate and type of secondary phases formed. The use of different tests to accelerate glass corrosion provides added confidence that the artifacts associated with each test method are understood and properly taken into account and that long-term corrosion behavior of the glass is understood. Three types of long-term experiments are recommended for the majority of the testing: 1) vapor hydration tests, 2) product consistency tests (PCTs), and 3) pressurized unsaturated flow (PUF) tests. Each of these tests and the alternative tests are discussed in detail in Section E-4.0.

E-3.2.2 Rate Law for Ion Exchange

Although ion exchange has been largely ignored in the recent literature on the glass-water reactions, the process has been the subject of numerous early studies. In fact, the traditional idea of glass "leaching" involves the basic mechanism of ion exchange in which an H^+ or H_3O^+ ion exchanges for an alkali ion (M^+) in the glass, thereby generating a hydrated layer on the glass surface. The overall chemical reaction describing the process can be written as:

$$\equiv\text{Si-O-M} + H^+ \rightarrow \equiv\text{Si-OH} + M^+ \tag{E.3}$$

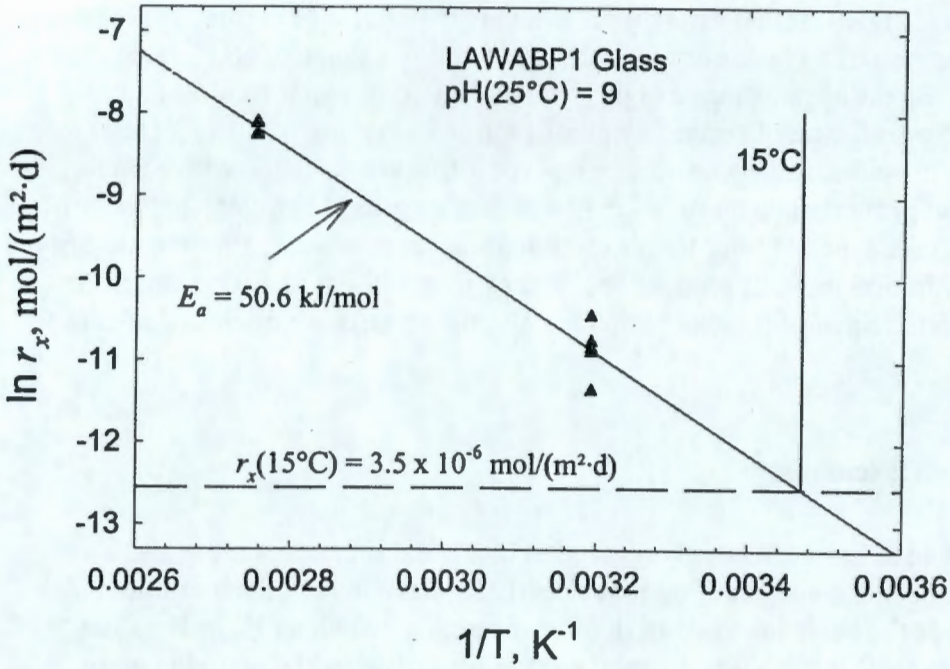
or

$$\equiv\text{Si-O-M} + H_3O^+ \rightarrow \equiv\text{Si-OH} + M^+ + H_2O. \tag{E.4}$$

Rana and Douglas (Rana and Douglas 1961a and 1961b) were among the first to report on this mechanism. Boksay, Bouquet, and Dobos (Boksay 1968) and Doremus (Doremus 1975; Doremus 1977; Lanford et al. 1979) pioneered the idea that ion counter-diffusion or interdiffusion is the rate limiting process for the exchange reaction. Recent data on a representative ILAW glass, LD6-5412, is also consistent with a diffusion-controlled release mechanism. Figure E-3.2 shows that Na is being released, presumably by ion exchange, at a linear rate with respect to $t^{1/2}$ in excess of the rate of matrix dissolution.

Detailed studies of the ion exchange process(es) are being conducted under the EMSP program. Once these studies are complete, an appropriate rate equation for ion exchange will be developed. The parameters that make up this model (possibly diffusion coefficients) will need to be determined from laboratory tests on the vendor glass compositions. However, until the EMSP studies are further along, it is premature to identify specific test methods and conditions.

Figure E-3.2. Excess Sodium Release via Ion Exchange as a function of Temperature for LAWABP1 Glass at pH(25 °C)=9.



E-4.0 LABORATORY TESTING

Waste glasses are laboratory tested to identify important corrosion processes under site-relevant conditions and to measure parameter values needed to model long-term behavior (see Section E-3.0). Three general classes of tests are proposed in this strategy: 1) characterization, 2) accelerated, and 3) service condition. Characterization tests are used to isolate and provide specific information about processes or parameters in theoretical models. Test conditions are usually very different from expected service conditions to highlight a particular process and minimize or hold constant other effects. Examples of such tests include measuring basic material properties, such as density and compressive strength. Characterization tests proposed in this plan related to glass corrosion are SPFT tests used to measure kinetic rate law parameters.

Accelerated tests are used to investigate corrosion behavior that will be important over the regulated service life of a disposal system within a laboratory time frame of a few years or less. Therefore, it is important to know likely site conditions over long times to determine what aspects of glass corrosion need to be considered in the performance assessment. Elevated temperatures and high glass surface area-to-solution volume ratio (S/V) often are used to accelerate the reactions and processes that lead to glass dissolution. It is important to verify that the technique used to accelerate a reaction or process does not cause a change in the rate-limiting step or mechanism of the process or, if it does, the change must be taken into account. For example, water diffusion, ion exchange, and hydrolysis will be affected to a different degree by changes in the temperature, and the overall temperature dependence of glass corrosion will include contributions from all three processes, although it may be dominated by one process under particular test conditions. Accelerated test methods that will be used to study the advanced stages of glass dissolution include the PCT, the vapor hydration test (VHT), and the PUF test. Tests will be conducted over a range of conditions to link the dissolution behavior under disposal conditions with that under accelerating test conditions.

Service condition tests are conducted to verify that the techniques used in accelerated tests do not change the alteration mechanisms. They are designed to approximate, to the extent possible in the laboratory, the physical and chemical environment of the disposal system. Because of the low temperature of the ILAW disposal system and the very slow rate of moisture flow expected in the disposal facility, laboratory tests approximating these conditions are unlikely to yield meaningful data in reasonable time periods. Consequently, no specific service condition tests are proposed in this plan. Rather, the parameter values for some tests are selected so as to reflect service conditions. For example, some PUF experiments may be performed with fractured glass monoliths instead of ground glass to provide service condition information on water flow paths and glass alteration processes in fractures. However, solution flow rate and temperature adopted for these tests will likely be higher than the expected conditions. Again, the purpose of the testing program is to demonstrate a scientific understanding of the processes controlling long-term glass corrosion so that the *models* describing these processes can be used with confidence in extrapolating to the expected service conditions.

Having described the general types of tests to be performed, the specific characterization and accelerated tests proposed under this plan are discussed in Sections E-4.1 through E-4.9.

E-4.1 SINGLE-PASS FLOW-THROUGH TEST

The SPFT test is an open system test where a solution at a known flow rate and constant temperature flows through a reaction cell that contains the sample. The configuration precludes recirculation of a portion of the effluent and so makes a "single-pass" through the reaction cell. Many different SPFT apparatuses have been developed, but these all can be classified under three basic types: 1) well-mixed batch, 2) packed bed, and 3) fluidized bed. The advantages and disadvantages of each design are discussed below.

E-4.1.1 Well-Mixed Batch Reactor

Figure E-4.1 shows a schematic of a typical batch flow through cell. Fluid and, optionally, a gas or gas mixture is pumped into the cell. Fluid exits the cell and is collected in a separate container for later chemical analysis. Mixing is accomplished by convection from the solution flow and (optionally) gas flow into the reactor. Some researchers have used a mechanical stirrer as well. The key advantages to using this type of reactor include the ability to use powdered or monolithic samples, use and control a gas or gas mixture during the test, and eliminate the need to control bubble formation in the fluid inlet, as is necessary with a packed bed reactor (see below). The only disadvantage to using a batch reactor is the need to ensure that when using powdered samples, sufficient mixing occurs to prevent agglomeration of the particles or formation of a stagnant solution around the sample.

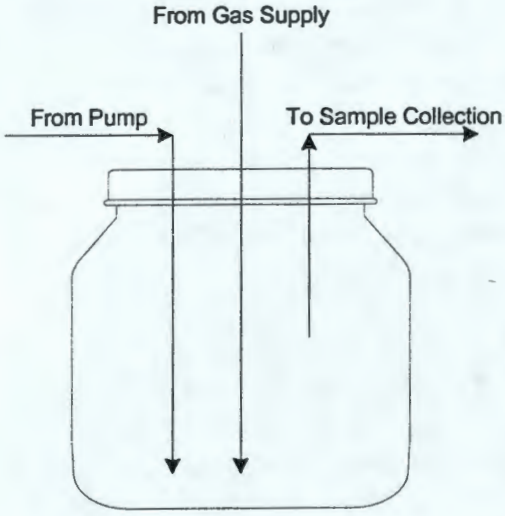


Figure E-4.1. Schematic of a Typical Batch Flow Through Cell.

E-4.1.2 Packed Bed Reactor

In a typical packed-bed type reactor, a porous bed of the test material is packed between two frits that have a nominal pore size smaller than the particle size of the sample. Solution flows through the porous bed and is collected in the same manner as with the batch reactor. The key advantage to using this type of reactor is that it eliminates the possibility of a stagnant solution layer as is possible with a batch reactor. However, this design has several important disadvantages. Care must be exercised to ensure that gas bubbles do not form in the inlet lines, which can become entrained in the porous bed. Entrained bubbles could exclude part of the

sample surface from contact with the fluid. McGrail and Olson (1992) reported that it was impossible to prevent bubble formation in their fluid inlet lines at 90°C, despite vigorous attempts to degas the solution. This limited the effective operating temperature to 70°C or less. The second disadvantage is that by the nature of the design, a concentration gradient will exist across the bed. However, depending on the specifics of the corrosion mechanism, this may or may not be of concern. Finally, the option of imposing a fixed gas partial pressure in the reaction cell is not available as it is with the batch-type reactor.

E-4.1.3 Fluidized Bed Reactor

The fluidized bed reactor is similar to the packed bed reactor except that much less sample is used so that, when sufficiently high flow rates are used, the sample particles are suspended or "fluidized" in the cell. The advantages of this design are that it eliminates the concentration gradient inherent with the packed bed reactor and ensures complete exposure of the entire particle surface area to the fluid. However, this test configuration has numerous disadvantages. First, depending on the particle size and density difference between the sample and the fluid, a relatively high flow rate is required to fluidize the bed. The flow rate required often can be sufficiently high that the solution exiting the reactor does not differ enough from background concentrations to be statistically significant. This is particularly problematic at lower temperatures because the flow rate required to fluidize the bed will change only slightly with temperature, but the dissolution rate of the test material may decrease by 2 to 3 orders of magnitude over the temperature range of 90 to 20°C, thus compounding detection limit problems. Second, the high flow rates required generate large volumes of effluent. This can present an expensive waste disposal problem, especially when testing radioactive materials. Third, because the particles are in motion and collide with one another, particle abrasion may cause a change in surface area during the test, which may be difficult to quantify. Fourth, the reactor must be carefully engineered and tested to prevent particles from settling in quiescent pockets and from transporting out of the reactor to the collection system.

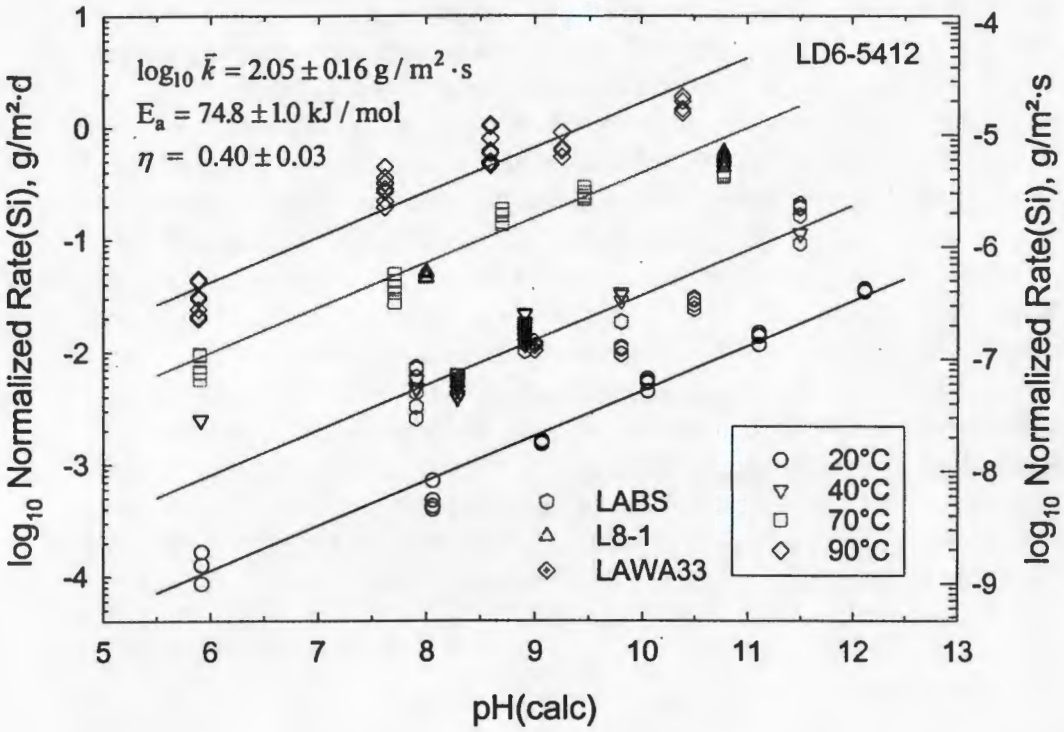
E-4.1.4 SPFT Testing Strategy

Based on the discussion of SPFT testing methods in Sections E-4.1.1 through E-4.1.3, the advantages of the batch reactor are sufficient to select it as the preferred test apparatus design. Testing must then be performed over an appropriate range of conditions. The pH of the water percolating through an ILAW disposal system will almost certainly be between pH 6 and 13, so the pH dependence of the rate must be determined over this range. Collecting SPFT data over the acidic pH range is not required. The activation energy (E_a) is another important parameter in the model, and SPFT tests must be run over a sufficient range of temperatures to accurately determine this parameter. A typical data set that provides the three parameters, k, E_a, and η, is shown in Figure E-4.2.

The procedures outlined in McGrail et al. (1997c) should also be followed in running SPFT tests over a sufficient range of flow rates to evaluate the parameters σ and K. Although K

should be independent of pH and perhaps σ as well, at least one additional experiment at another pH should be performed for confirmation. Finally, test results (McGrail et al. 1997c) on LD6-5412 glass indicated the possibility that an additional element (besides Si), such as Al, may need to be included in the affinity term. If a similar conclusion is reached based on the SPFT test results for the vendor glass compositions, then a series of tests should be performed where the concentrations of the test element(s) are varied independently. An analytical solution for a mixed Si-Al kinetic rate ILAW model was developed by McGrail et al. (1997c) and can be used to develop a statistical design for these tests.

Figure E-4.2. SPFT Test Results for Various ILAW Glasses.



E-4.2 VAPOR HYDRATION TEST

Figure E-4.3. Schematic of VHT Test.

The VHT is the simplest accelerated test to be used in the recommended testing strategy because there is only one degree of freedom in test parameters that can be varied, temperature, excluding test duration and humidity. The principle use of the test is as a convenient means of generating alteration phases for analysis within a short period, but it also is

useful as a qualitative measure of the effect of alteration phase formation on the corrosion rate. The VHT is a static test in which a monolithic sample is exposed to water vapor in a sealed vessel, as illustrated in Figure E-4.3. At relative humidity (RH) above about 80%, a thin film of water condenses on the sample. The thickness of the film increases with the RH, and it is in this film that the glass corrodes. The thickness of the film determines the effective S/V ratio of the system; for a uniform layer, the S/V ratio is simply the inverse of the thickness of the film. Most VHTs have been performed in 100% relative humidity at temperatures above 100°C, although tests have been conducted at lower humidities and temperatures. At the completion of a test, the sample is removed from the vessel, and the reacted surface is analyzed with high-resolution electron microscopy. Discrete precipitated crystalline phases usually form when the sample corrodes. The solution evaporates from the sample when the test is terminated and is not available for analysis. This precludes using the test to evaluate and test models of glass corrosion behavior, which require detailed measurements of the solution composition in contact with the glass.

Vapor hydration has been used by archaeologists to replicate the weathering observed on obsidian artifacts recovered from terrestrial sites (Friedman and Long 1976). The method has also been used to simulate the long-term weathering of tektite (Mazer et al. 1992) and basaltic glasses (Byers et al. 1986). Because the test method produces the same alteration phases that form during the weathering of natural glasses over long periods, the test has also been used to accelerate the corrosion of waste glasses. High-level waste glasses commonly form clay layers in VHTs. Glasses having compositions relevant to anticipated ILAW for Hanford formation zeolite phases similar to analcime, gobbinsite, and phillipsite.

To extract kinetic information, VHTs will be run over a minimum of four temperatures, all at 100% RH. However, it is important to recognize that the rate measured in VHTs is not the glass dissolution rate, but the rate at which alteration phases are formed. It is presumed that precipitation rates of the alteration phases are much faster than the glass dissolution rate. Tests with radioactive glasses will be used to measure the distribution of radionuclides between alteration phases and residual phases (see Section E-4.6).

E-4.3 PRODUCT CONSISTENCY TEST

The PCT has been standardized as an ASTM standard procedure (ASTM 1994). The ASTM standard includes two methods: PCT Method A was developed specifically for verifying process control of vitrified HLW forms and is conducted with specific values of test parameters; PCT Method B does not specify the values of test parameters. Because the PCT Method B encompasses commonly used variations of test parameters, we refer to all static dissolution tests with crushed glass generically as PCTs.

The PCTs are conducted by reacting an aliquot of crushed glass that has been sieved to isolate the desired size fraction with an aliquot of a solution in a sealed vessel at the desired temperature. The glass S/V can be calculated from the glass-solution mass ratio and the specific

surface area of the crushed glass to determine the average dissolution rate during the test. Tests are conducted as batch tests wherein a separate test is run for each duration. At the end of the test, the solution is analyzed for pH and the concentrations of dissolved glass components. The latter are used to calculate the amount of glass that has reacted and the dissolution rate. The reacted glass surface also can be analyzed to help characterize the reaction mechanism, such as when an alkali-depleted layer or alteration phases form.

Because the test conditions in PCTs are water-saturated and static (closed-system), which are not consistent with expected disposal system conditions, including a significant number of PCTs in this plan requires explanation. Traditionally, PCTs are used as a means to study advanced stages of corrosion at high glass-solution mass ratio (or S/V), high temperature, and long test duration. It is argued that the reaction conditions in the PCTs are similar in several regards to the conditions expected in a disposal site. Although waste glasses will not be crushed, the S/V values commonly used in PCTs are similar to the effective S/V values when a small volume of water contacts a glass. The static conditions of the PCTs are also probably not too different from the very low water infiltration rates anticipated. Therefore, PCTs probably provide a fair simulation of the corrosion behavior after a small amount of water has contacted a waste package and reveal key reaction processes. While each of these statements is true to some extent, they provide insufficient justification for running PCTs as they establish only heuristic connections with modeling.

Consistent with the model-centered approach used in this plan, the PCT is used to calibrate the substantial set of supporting geochemical data that are required for modeling the dynamic evolution in solution chemistry that occurs as a consequence of glass-water reactions. The term "calibration" in this context refers to a complex iterative process whereby the evolution in solution composition and secondary phase formation observed in PCTs is reproduced, with a reasonable level of uncertainty, in a geochemical model of the system. The process is complex because during a closed-system test, like the PCT, changes in the concentrations of dissolved glass components, the solution pH, and, in some cases, dissolved air components must be modeled simultaneously. Fortunately, the geochemical simulator selected for modeling this system, the EQ3/6 code (Wolery and Daveler 1992), has capabilities for handling these complexities.

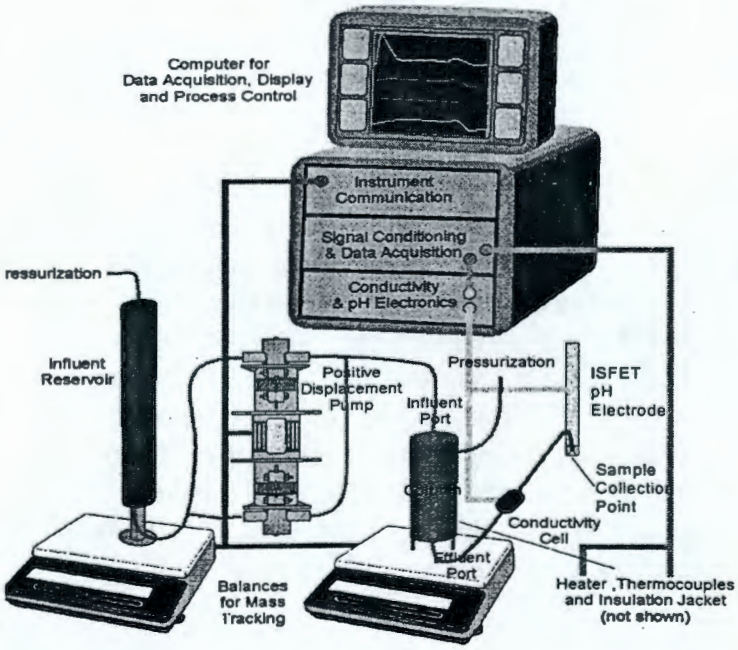
The calibration process requires, among other things, estimating solubility product values for secondary minerals that are identified in the PCTs, but for which thermodynamic data are not available. These values can be generated by fitting to the PCT data or by using empirical methods, such as with a polymer model (Mattigod and McGrail 1998). However, if a particular phase is found to have an important effect on long-term glass corrosion rates, it is recommended that independent measurements of its solubility product be performed. Once the geochemical model has been calibrated against the PCT data, the significant aqueous speciation and dissolution-precipitation reactions in the model can be identified. This reaction set and supporting thermodynamic data then make up the reaction network that is used in the reactive transport model to compute radionuclide release from the disposal system (Bacon and McGrail 1997).

Because the dissolution behavior of glasses in PCTs are usually strongly affected by the affinity term, PCT results can also be used to measure of the values of the saturation concentration (K) and the net reaction order variable (σ). However, the effect of a changing pH and solution composition complicates the extraction of these values. Instead, the results of PCTs conducted for long durations can be used to confirm the values of K and σ that are derived from the SPFT tests (see Section E-4.1.4).

There are two degrees of freedom for parameter variation in PCTs, temperature and S/V, excluding test duration. It is recommended that PCTs be performed at a minimum of four separate temperatures between 20 and 90°C and at two S/V ratios. Tests at elevated temperatures are needed to measure the trends in the corrosion behavior as a function of temperature and to determine if the corrosion mechanism changes with temperature. Use of higher S/V ratios provides another means of accelerating the extent of reaction. However, there are limits to the values of S/V that are practicable. At S/V ratios >20,000 m^{-1}, test artifacts become important. Initial alkali ion exchange and dissolution of fines can increase leachate ionic strength and pH to values that are unrealistic for a disposal system subject to open-system mass transport. In PCTs conducted in containers impermeable to air, the available $CO_2(g)$ can also be consumed, which compounds the pH excursion from the ion-exchange and limits the formation of important carbonate minerals. Consumption of water from hydrolysis of the glass becomes an important consideration, as the S/V could be a steeply time-varying function. This makes interpretation of the test data much more difficult. Consequently, it is recommended that the S/V be limited to a maximum of 20,000 m^{-1}.

E-4.4 PRESSURIZED UNSATURATED FLOW TEST

Figure E-4.4. PUF Apparatus.



The PUF test is a new technique developed at PNNL for testing waste forms (McGrail et al. 1997a and 1997b). The experimental design provides a novel way to study waste form corrosion behavior under unsaturated conditions, subject to open-system flow and transport. Like the SPFT test, the PUF test provides three degrees of freedom for parameter variation: temperature, glass surface area, and flow rate.

The basic test apparatus consists of a column packed with glass particles (or other material) of a known size and density, and a computer data acquisition and control system (see Figure E-4.4). The

column is fabricated from a chemically inert material so that dissolution reactions are not influenced by interaction with the column. A porous titanium plate of proprietary design is sealed in the bottom of the column to ensure an adequate pressure differential for the conductance of fluid while operating under unsaturated conditions (Wierenga et al. 1993). Titanium was chosen because it is highly corrosion resistant and has excellent wetting properties. When water saturated, the porous plate allows water but not air to flow through it, as long as the applied pressure differential does not exceed the air entry relief pressure or "bubble pressure" of the plate. The computer control system runs LabVIEW™ (National Instruments Corporation) software for logging test data to disk from several thermocouples, pressure sensors, inline sensors for effluent pH and conductivity, and column weight from an electronic balance to accurately track water mass balance and saturation level.

Several important findings have resulted from PUF experiments that have been performed on LD6-5412 glass and SRL-202 glass, a reference HLW glass for the Defense Waste Processing Facility. With LD6-5412 glass, precipitation of zeolitic secondary phases was correlated with an acceleration of the glass reaction rate, exactly as was found in PCTs conducted at a high surface area-to-volume ratio (S/V) of 20,000 m^{-1} and the same 90°C temperature. However, in the PCTs, the accelerated reaction stage did not occur before about 120 to 240 days. In the PUF test, the accelerated reaction stage was reached after only about 12 days. This is an acceleration factor of about 10 to 20, even though the specific surface area of the glass used in the PUF test was about 10 times smaller than the specific surface area of the glass used in the PCTs. In the PUF test with SRL-202 glass, precipitation of an aluminosilicate phase, pianlinite [$Al_2Si_2O_6(OH)_2$], was correlated with a 50% decrease in volumetric water content, indicating a change in the hydraulic properties of the sample. This is the first evidence ever reported that corrosion and secondary phase precipitation can induce changes in the unsaturated hydraulic properties of a glass waste form. Experiments with Pu-containing glass packed in a thin layer between two layers of SRL-202 glass (not containing Pu) have also shown that >90% of the Pu, brought into solution from corrosion of the Pu glass, exits the column as a colloid. This is important because to exit the column as a colloid in this experiment, the colloids must have been transported approximately half the column length through a layer of crushed SRL-202 glass. Consequently, the PUF test provides a means to study the formation and transport of radiocolloids under unsaturated flow conditions.

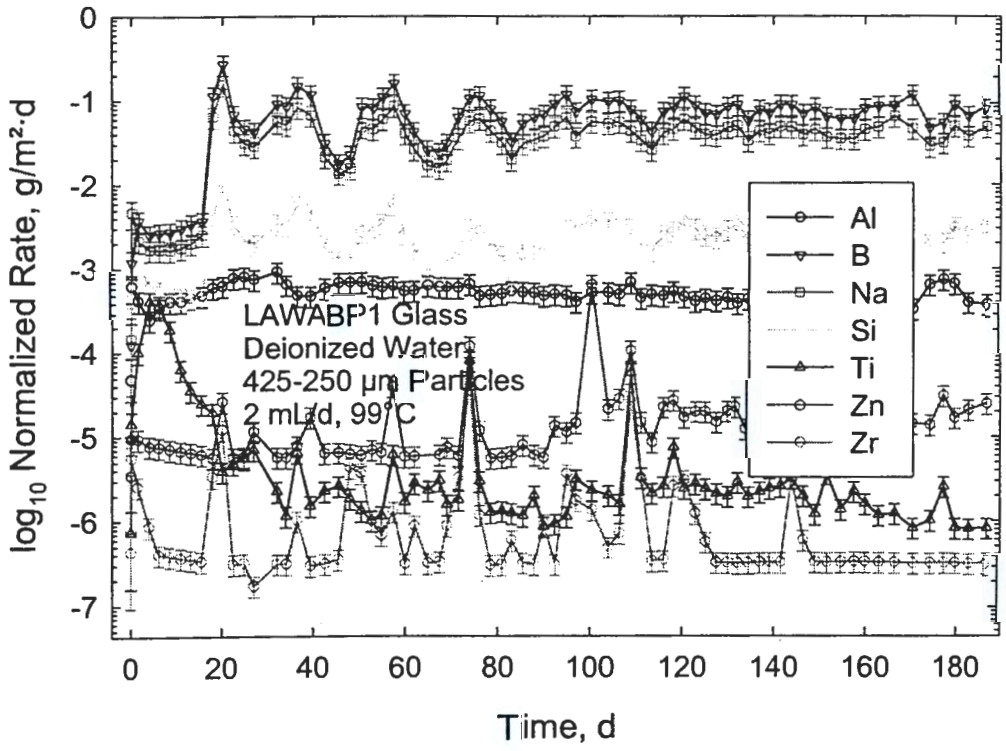
Based on the findings from the PUF experiments that have been run to date, the test will play an important role in the overall testing strategy. First, because the test combines glass corrosion, unsaturated flow, and mass transport processes under controlled conditions, it is ideally suited to provide data for validation of the reactive transport model being used for long-term performance calculations for the disposal system (Bacon and McGrail 1997). Second, the demonstrated ability to accelerate the transformation of the glass into alteration products means that the test will provide critical information on these phases, and the stability of the glass with respect to the formation of these phases, in relatively short periods of time. Third, aged samples from the test can be independently evaluated for changes in unsaturated hydraulic properties, thereby providing critical data for modeling fluid flow through the disposal system. Fourth, the flow-through configuration allows for different engineered-barrier materials to be tested in

combination to evaluate interactive effects on glass corrosion rates and/or radionuclide release (see Section E-4.7).

Although the PUF test is assigned a key role in the overall testing strategy for the ILAW program, it must be recognized that because of the complex apparatus required, the number of PUF experiments that can be performed is constrained by relatively high costs per test, equipment availability, and access to skilled technicians to run the test. Consequently, it is recommended that PUF tests be performed at three different temperatures only. Also, because very little information currently exists on the effect of flow rate on the PUF test results, a series of tests should be performed on one vendor glass formulation that covers a range of flow rates. These experiments should be designed to examine whether the corrosion rates can be correlated with a reduced parameter, such as the ratio of volumetric flow rate to glass surface area, a well-defined correlating parameter in SPFT tests (McGrail et al. 1997c). Finally, tests are recommended with radioactive glasses (see Section E-4.6) and a series of materials interaction experiments with at least cement and glass combinations, but perhaps including iron or iron corrosion products as well (see Section E-4.7).

Typical results from the PUF test are shown in Figure E-4.5.

Figure E-4.5. Normalized Release Rates in PUF Tests with LAWABP1 Glass.



E-4.5 OTHER TEST METHODS

Other test methods have been used to measure the values of model parameters and to study glass corrosion behavior under site-relevant conditions. Some of the more commonly used test methods are described below. The reasons why these tests are not included in the testing strategy for Hanford Site ILAW are also discussed. Basically, the other test methods described below provide little or no additional relevant information when compared with the tests proposed in Sections E-4.1 through E-4.4.

E-4.5.1 Soxhlet Tests

Soxhlet tests provide a method for measuring the glass corrosion rates under highly dilute conditions. Soxhlet tests are performed by reacting a sample with condensed fluid in a refluxing apparatus. Water is boiled from a reservoir, condensed in a reflux tube, then allowed to drip into a small cup that contains the sample. The solution in the sample cup is refreshed as condensate drips into the sample cup. Complete volume replacement of the fluid in the cup typically occurs every few minutes but the precise replenishment time depends on the sample cup volume and refluxing conditions. The refluxing action provides an effective flow rate that can be varied by controlling the applied heat flux to the device. Tests are usually run to attain a flow rate sufficiently high that the solution in the sample cup remains highly dilute. The corrosion rate can be measured by periodically removing a sample of the solution in the reservoir for analysis. The Materials Characterization Center Soxhlet test (MCC-5) specifies the use of a monolithic sample and an all-Teflon apparatus (Strachan et al. 1981), although crushed samples and stainless steel apparatuses have also been used.

Soxhlet tests have been used to measure the initial corrosion rate of glasses as a function of the temperature, particularly for European HLW glasses (Delage and Dussossoy 1991; Tovena et al. 1994). Unfortunately, because the system operates with condensed vapor, the dissolution rate can only be easily measured at the pH of distilled water at the test temperature. Because the dissolution rate is required over a pH range of 6 to 13, the SPFT technique (see Section E-4.1) is preferred for these measurements.

E-4.5.2 MCC-1 Test

The MCC-1 static leach test method was developed to compare the durabilities of candidate waste forms developed to stabilize high-level nuclear waste (DOE 1982). The test procedure has been standardized by the ASTM (ASTM 1992). The method calls for placing a monolithic specimen of known geometric surface area into a volume of solution such that the S/V is 10 m^{-1}. The test may be performed with demineralized water, a reference silicate solution, or a reference brine. The reference temperature and time are 90°C and 28 days, although

temperatures of 40 or 70°C and other durations can be used.[a] The MCC-1 test typically provides a solution-dominated system in that the leachate remains dilute as the glass dissolves. However, tests conducted for long time periods may be affected by changes in solution chemistry. Samples corroded in MCC-1 tests show details of the chemical and physical alteration of the glass surface and have provided insight into mechanisms controlling the initial stages of corrosion (Bates et al. 1991; Oversby and Phinney 1992).

While the MCC-1 method was originally designed to compare the relative chemical durabilities of candidate waste forms, the test can be used to characterize several aspects of the corrosion process in conjunction with a corrosion mechanism. Tests conducted for short time periods provide a simple means of measuring the glass corrosion rate under dilute conditions. Tests have been performed in buffer solutions or solutions spiked with various glass components to determine the effects of the leachate chemistry on the glass corrosion rate (Advocat et al. 1991). Longer term tests can be run to monitor the alteration of the glass surface during corrosion to investigate the corrosion mechanism (Strachan 1983; Bates et al. 1991). However, because each of these processes is covered in greater depth using SPFT and PCTs, there is no compelling need to include MCC-1 tests in the testing strategy for ILAW glass.

E-4.5.3 MCC-3 Solubility Test

The MCC-3 test was designed to measure the maximum solubility of a waste form in the solution of interest (MCC 1984). This test method formed the basis of the PCT. Tests were to be conducted until the solution composition did not vary with the reaction time (i.e., until the glass "saturated" the solution). However, application (Shade and Strachan 1986) of the test method showed that 1) constant solution compositions were not achieved within a few weeks, and 2) the solution composition depended on the particle size of the glass used. The first observation is a direct result of the glass reactivity, while the second is due to the different S/V that results from the different total surface areas of different sieve fractions. Tests conducted at higher S/Vs usually generate more concentrated solutions. The MCC-3 procedure was subsequently modified such that 1) a single size fraction is specified, and 2) the test vessel is continuously agitated during the test, usually by placing the vessel on a roller. The PCT has effectively replaced the MCC-3 test in most laboratories, although all of the complications associated with the use of crushed glass found during development of the MCC-3 tests are common to the PCT procedure.

E-4.5.4 Periodic Replenishment Tests

Several test methods have been developed in which leachate solution is periodically removed from an ongoing static test and replaced with an equal volume of fresh solution. Such replacement tests have been used to simulate very low flow rates that cannot be attained using

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[a]The MCC-2 test procedure is a variation of the MCC-1 procedure that permits reaction temperatures of 110, 150, and 190°C.

mechanical pumps (Barkatt et al. 1983 and 1984). Either monolithic or crushed samples can be used, and different starting solution compositions can be used. Specific test methods have different replacement schedules and replace different fractions of the total solution volume. For example, the test designated by the International Standardization Organization (ISO) calls for replacing the entire solution volume daily for the first week, every week for eight weeks, monthly for six months, and twice yearly thereafter (Hespe 1971). The American Nuclear Society test method ANS 16.1 and variations of the ANS 16.1, such as the Dynamic Leach Test (DLT) and the Accelerated Leach Test (ALT), are similar total volume exchange tests. They differ primarily in the replacement schedules. These tests were developed to characterize materials from which contaminants are assumed, a priori, to be released by a diffusion-controlled process, such as with grouts and cements. Although the ANS 16.1 test is required for ILAW in the Phase 1 Hanford Privatization Contracts, this type of test provides little insight to the long-term corrosion behavior of glass waste forms. This is because concentrations of the waste form components and solution pH change over time in the fluid contacting the waste form until the fresh fluid is injected, which then causes a sudden and abrupt change in the solution chemistry. Such conditions are extremely difficult to interpret and cannot be treated adequately with conventional geochemical computer codes. Consequently, no periodic replenishment tests are proposed in this plan.

E-4.5.5 Unsaturated or "Drip" Test

The Unsaturated Test or "drip test" was developed at ANL to simulate the corrosion of a waste glass that is intermittently contacted by transient water in an unsaturated geologic environment, such as that at Yucca Mountain (Bates and Gerding 1986). A sample of the waste form and possibly other engineering or geologic materials is placed in the center of a reaction cell. A small amount of liquid water is injected through a septum and allowed to drip from the feed tube onto the upper surface of the sample. Initial aliquots of added water are vaporized until the air is saturated; subsequent aliquots drip onto the sample. Solution that drips from the sample during the test and collects in the bottom of the test vessel can be analyzed to quantify the amount of glass that has dissolved and been transported away from the waste form. The corroded glass can also be analyzed to assess the corrosion mechanism, study materials interactions, and identify any alteration phases that form (Woodland et al. 1991). Unsaturated tests are being used as one of the primary sources of radionuclide release data in a total system performance assessment for the Yucca Mountain Site (Stout and Leider 1996).

Unfortunately, the "drip test" has several drawbacks with respect to evaluating ILAW glass behavior relevant to disposal at the Hanford Site. First, the drip test is a quasi-static method, similar to the periodic replenishment tests discussed in Section E-4.5.4. Consequently, the same difficulties exist in applying conventional modeling tools for modeling the experiment as were previously described for these tests. Second, the exact amount of fluid in contact with the sample at any given time is not known (and likely variable), so the effective S/V, a key parameter needed in modeling, is poorly defined. Finally, the drip test was originally designed as a service condition test corresponding to a particular scenario at the Yucca Mountain Site where fluid may periodically drip on the waste packages from fluid-filled fractures. This "drip" scenario does not represent a possible hydrodynamic condition for the ILAW disposal system,

which is located in a shallow, sandy soil. The PUF method is preferred for unsaturated testing because it applies a steady unsaturated hydraulic condition on the sample, and the total amount of water in contact with the sample is continuously monitored.[a] These features make the PUF test amenable to modeling using standard computational methods and tools.

E-4.5.6 Accelerated Dissolution Test

The accelerated dissolution test (ADT) was designed to measure the dissolution rate of a glass in the presence of its alteration phases and in a solution that is in near equilibrium with those phases. The alteration phases are generated by vapor hydration of a sample of the glass being evaluated, and the nearly saturated solution is generated in a PCT conducted at high S/V and at the temperature at which the dissolution rate is to be measured. The dissolution rate is measured by mass loss of a fresh monolithic sample. To date, the ADT has only been applied to measure the dissolution rate of the LD6-5412 glass (Ebert et al. 1996). The rate measured with ADTs was consistent with that estimated from long-term PCTs and that extrapolated from VHTs. However, an inoculation period existed in which the dissolution rate was at first very low and then increased because of the effects of the alteration phases. The occurrence of this period, which was of different durations in replicate tests, is not fully understood, but may be related to time needed for the solution and alteration phases to equilibrate. A specific role for the ADT has not been identified in this plan, but the test will likely continue to be used periodically to confirm long-term dissolution rates measured in VHTs and PCTs.

E-4.6 REQUIRED TESTS WITH RADIOACTIVE GLASSES

In Sections E-4.1 through E-4.5, no distinction was made with regards to testing "cold" or fully radioactive glasses. Because the ultimate goal of the testing program is to evaluate long-term radionuclide release rates, and not glass corrosion rates per se, some testing must obviously be performed with radionuclide-containing materials. However, it is not required that all testing be performed with radioactive materials. This is because the radionuclides found to be of most concern in the performance assessment (Mann et al. 1997) for the ILAW disposal system are ^{99}Tc, ^{79}Se, and ^{129}I. These radionuclides are extremely soluble under the oxidizing conditions that are expected at the site. Consequently, the rate of glass corrosion ultimately determines the rate of release for these important elements. However, at very long times, Pu and U isotopes also contribute to the computed doses from the groundwater pathway (Mann et al. 1997). These elements are expected to form sparingly soluble precipitates in the disposal vault and also may be transported as colloids, so the release rates will not depend directly on the corrosion rate of the glass. For these elements, experiments with radioactive glasses are needed to identify the solubility-controlling phases. Release rates of the soluble radionuclides also need to be checked

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[a]The capability to monitor spatial variations in moisture distribution during a PUF test will be available shortly at PNNL with the installation of an X-ray computed microtomography system.

for congruency with the rate of glass corrosion to eliminate the possibility of a mechanism, such as ion exchange, that could selectively enhance release rates.

Because only very limited data have been obtained with a single ILAW glass (LD6-5412) doped with ^{99}Tc, it is difficult at this time to define what amount of testing with radioactive glasses is required. The level of testing also depends strongly on future funding levels for the ILAW disposal program. A reasonable conjecture includes a reduced set of VHT, PCT, and PUF tests with a fully radioactive glass containing each radionuclide of interest. The reduced set may include VHTs at one temperature, PCTs at two temperatures but only the highest S/V, and PUF experiments, also at two temperatures, but at two flow rates and one glass surface area. Concentrations of the radionuclides in the test glasses likely should not reflect the expected activity levels in the actual glass product. Consideration should be given to doping the test glasses at significantly higher levels than expected so that it is easier to identify secondary solids containing the radionuclides. Because the radionuclide concentrations and radioactivity levels of ILAW glass are very low, there is little or no concern regarding radiolysis-induced experimental artifacts or having the radioactive glasses exhibit different long-term corrosion behavior when compared with nonradioactive glasses.

E-4.7 MATERIALS INTERACTION TESTS

The ILAW disposal system will include the following additional materials, besides glass, that are used in construction or as part of the waste package: 1) concrete used in the vault walls, ceiling, and floor; 2) carbon or stainless steel containers used to hold the glass; and 3) filler material used inside the containers to consume void space. Of these materials, the concrete is of the most interest because of the anticipated volume required, approximately 320,000 metric tons (Burbank 1997), and the chemical effect of the concrete in increasing pH and Ca, Na, and K concentrations. Corrosion of the metal containers will generate iron oxyhydroxides that are known to adsorb or coprecipitate with Si (Harder 1978). This can act as an additional sink for Si, resulting in enhanced rates of glass corrosion (McVay and Buckwalter 1983; McGrail 1986). However, iron hydroxide precipitates also may have a beneficial effect in that they are well known adsorbents for a wide variety of metals, and so could lower release rates for some radionuclides. No specific filler material has been identified, so the significance of this material to glass corrosion behavior cannot be assessed at this time.

Because the concrete is used exclusively on the exterior of the vaults, the interactive effects of the concrete on the glass (if any) will most likely be confined to a region at the "interface" between the two materials. This is because diffusion and/or advection limit the total mass flux of OH^-, Ca^{2+}, etc. from the concrete that is available for reaction with the glass. To properly test this mass-transport rate-limited process in the laboratory, a technique is required where mass transport by diffusion and/or advection from the concrete into the glass can be controlled. Batch experimental methods, such as the VHT or PCT, are intentionally designed to be well-mixed systems to avoid mass transport constraints on the reaction processes. Consequently, these methods are unsuited for this materials interaction study. Fortunately, the

PUF technique provides a means to control mass transfer and transport rates by the particle size and flow rate used in the test. The concrete and glass can be configured sequentially, such that the water can "equilibrate" with the concrete before entering the porous glass bed. By comparing differences in corrosion rates and radionuclide release, with and without the concrete, and by examining the alteration products formed, especially at the interface between the two materials, a good understanding of the interactions between the materials can be developed.

Interactive effects with the container or iron corrosion products are more difficult to assess because the material will be distributed throughout the disposal system, including the interior region of each vault. In this case, the mass transport limitations discussed above for the concrete do not apply, at least on a scale larger than a single waste package. A highly localized assessment of the interactive effect could be obtained by running a VHT where the container material and glass are "sandwiched" together. An alternative would be to run PCT or PUF experiments where the glass and container material particles are mixed together in volume percentages equivalent to what is expected in the current disposal system design. Glass corrosion rates would then be directly compared with and without the metal present to obtain a direct assessment of the interactive effects, if any. Care must be exercised in these tests to ensure that the available O_2 is not consumed from oxidation of the steel.

E-4.8 ROLE OF NATURAL ANALOGS

Analogy is a scientific method that uses inference from certain resemblances to imply a probable further similarity (Ewing and Jercinovic 1987; Petit 1992). The term natural analog refers to a material or process that resembles those expected in a waste disposal system or the methodology used to study and assess them. The analysis of analog materials has been used to enhance confidence in the validity of long-term predictions made with mechanistic models, as well as assisting in the development of those models (ASTM 1991). By studying the alteration of natural materials that has occurred over millions of years due to weathering in a range of terrestrial environments, insight has been gained into the long-term corrosion behavior of waste glasses in a disposal environment. This is done by characterizing the natural alteration of the analog material and comparing it to alteration produced in laboratory tests with naturally occurring or chemically identical synthetic materials. In addition to studying the natural alteration of a material, it is also important to identify the alteration mode(s) that controls corrosion of the analog material and to verify that the same process that controls the rate in the short-term testing also controls the long-term behavior that is being modeled. Applications of natural analogs include

- comparing alteration products generated during long-term corrosion of natural glasses to those generated in accelerated laboratory tests to verify experimental methods of accelerating glass corrosion

- studying specific reaction processes that are important in waste glass corrosion and are highlighted in the corrosion mechanism of a natural material

- studying the relationship between the kinetics of natural glass corrosion measured in short-term laboratory tests and the known long-term durability of the glass to verify that the appropriate reactions and processes are modeled for long-term calculations

- calibrating the degree to which a laboratory test accelerates a corrosion mode.

The use of natural analogs to extract kinetic information is usually limited to bounding estimates because of the uncertain and changing conditions to which the analog material has been subjected, including wet-dry cycling, variations in temperature and water composition, etc., and uncertainty regarding the age of the sample itself. In most cases, kinetic information must be extracted by examining reacted solids that are usually incomplete assemblages of alteration phases. Nevertheless, corroded natural glasses provide valuable insight regarding corrosion processes that are important to long-term corrosion and confidence in the accelerating properties of laboratory tests.

Geologists have studied samples of basaltic glasses recovered from marine, geothermal, and subglacial environments. Corrosion of basaltic glasses results in the formation of palagonite, which is a generic term for an assemblage of alteration phases that includes clays and zeolites. The same phases have been observed to form in nature and in laboratory tests (Lutze 1985; Byers et al. 1985). In both natural settings and in laboratory tests, the amounts and in some cases the identities of phases that are formed depend both on the glass composition and the environmental conditions (e.g., the temperature, chemistry, and volume of the solution contacting the glass). Waste glasses have been shown to react similarly to basalt glasses in laboratory tests and to transform to many of the same phases. This provides evidence that 1) the laboratory test accelerates the corrosion behavior of basalt glasses and leads to the same phases formed in nature over very long time periods and 2) the same processes that control long-term basalt glass corrosion in terrestrial sites (and under terrestrial conditions) also control the long-term corrosion of waste glasses.

Long-term testing of natural analog glasses using VHT and PCT methods is being funded at ANL through separate DOE programs. Consequently, there are no current plans to support additional VHT and PCT testing of natural analogs through the ILAW disposal project. However, the PUF method has only recently been developed, and no natural materials have been examined using this test method. Consequently, a small number of long-term PUF tests should be performed using a selected natural analog. The preferred material would have a much higher alkali content than is typical of rhyolitic or basaltic glasses and would have been subjected to environmental weathering conditions analogous to those expected for the disposal system. One possible site for such samples is the obsidian flows at Newberry Crater, Oregon (25 miles south of Bend, Oregon), which range in age from 1,200 to about 6,700 years B.P. This area has an arid environment similar to that of the Hanford Site and potentially provides similar analogs at different ages within a short distance of each other. Regardless, the selected natural analog sample for study must be as well-characterized as possible in terms of the sample age and weathering conditions if it is to be useful for comparing against data from PUF experiments.

E-4.9 LABORATORY TESTING STRATEGY SUMMARY

Table E-4.1 represents a compilation of the recommended testing discussed in Sections E-4.1 through E-4.6. To the best of the authors' current knowledge, this is the minimum matrix of laboratory tests required to fully characterize the long-term corrosion and radionuclide release behavior of a glass waste form for disposal at the Hanford Site. A duration limit of 2000 days was selected because the major decisions regarding acceptable glass compositions and disposal system design need to be made in approximately the next 5 years. Consequently, tests beyond this time frame will have little impact on the overall program.

A key issue not discussed in the strategy up to this point is variability. Variability in the waste stream delivered to the private vendors will result in variability in the composition of the glass product; i.e., no single glass composition represents the entire inventory of glasses to be produced during the vitrification campaigns. Glass compositional variability must also be addressed in evaluating the long-term performance of the disposal system. One approach is to identify a selected set of glasses that represent compositional extremes and then perform a reduced set of experiments on these glasses. The results from these tests can then be used to bound the release rate behavior for entire compositional space. Once data are available on the reference vendor glass compositions and the variability expected during processing is quantified, specific compositions and tests can be proposed.

Table E-4.1. Master Test Matrix for Long Term Performance Evaluation of a Glass.

Test Method	Temp	pH	Flow Rate	Surf. Area	Sol. Vol.	Duration Range (days)
SPFT(k, E_a, η)	4	6	Var.[1]	1		14-30
SPFT(K, σ)	1	2	15	1		14-30
VHT	4					1-2000
PCT	4			2[2]	1[2]	30-2000
PUF	3		3	2		30-2000
Rad. Glasses						
VHT	1					1-365
PCT	2			1[2]	1[2]	30-2000
PUF	2		2	1		30-2000

[1]Flow rate sufficiently high so that $Q/K \approx 0$.
[2]Varied as S/V.

E-5.0 FIELD TESTING

Field testing provides a unique opportunity to obtain site-relevant data on the corrosion behavior of waste forms under conditions that more closely approximate service conditions than is possible in the laboratory. Recognizing this fact, several burial studies with glass waste forms have been conducted in the United States and abroad. One of the first studies involving glasses was carried out in the early 1980's at the Stripa Site in Sweden. The Stripa Site is an abandoned iron mine located in a granitic formation approximately 350 m below the surface. Specimens were fabricated in a "pineapple slice" geometry and stacked into assemblies that were either heated to 90ºC or maintained at ambient mine temperatures (8 to 10ºC). Samples were extracted at predetermined intervals (0.25, 0.5, 1, and 2 years) for extensive characterization using surface spectroscopic and microscopic analytical techniques. In the United States, a comprehensive field test involving HLW glasses was performed for the Materials Interface Interaction Tests (MIITs) at the Waste Isolation Pilot Plant (WIPP) in Carlsbad, New Mexico. Located approximately 650 m below the surface in the Salvo salt formation, a wide array of glasses and waste package materials was tested for 5 years. Pineapple slices of test samples were stacked on heated Teflon assemblies in such a manner as to test a variety of interfacial reactions. The United States also has participated in burial studies conducted at the Mol Site in Belgium (clay geology) and Ballidon Site in the United Kingdom (limestone) in an effort to test the performance of a simulated HLW glass (SRL-165) in other geologies.

At the Hanford Site, grout waste forms have been field tested in lysimeters at the Grout Waste Test Facility located in the 300 Area (Last et al. 1995). A lysimeter essentially consists of a corrugated steel container buried flush with the ground surface. Lysimeters can range in size from small soil-filled cans, a few centimeters in diameter and a few centimeters deep, to large caissons that are several meters wide and tens of meters deep. Lysimeters can be simply designed with little or no peripheral instrumentation, or they can be designed with extensive monitoring features, including devices for monitoring temperature, water content and matric potential and devices for extracting samples of water for chemical analysis. Lysimeters have been built and used at the Hanford Site for a variety of reasons, including the assessment of recharge rates, biointrusion studies, radionuclide transport studies, evapotranspiration studies, and field-scale waste-form performance tests (Gee and Jones 1985; Rockhold et al. 1995).

Although lysimeters have several disadvantages, they are a logical choice for field testing of the glass corrosion and contaminant transport models being developed for the ILAW disposal system. The devices provide a way to combine an ILAW glass, Hanford formation soil, and perhaps other engineered materials in a well-controlled test, but on a scale that is not practicable in the laboratory. The test is controlled because the walls of the lysimeter form a physical boundary in the system being studied that defines a fixed volume for calculating water storage and tracer mass balance and restricting the geometry of flow within the lysimeter to essentially one dimension. Interpretation of complex temporal variations in tracer release and transport is much simpler in one spatial dimension. The field scale affords the opportunity to monitor contaminant release and transport in time and space that is not possible in laboratory experiments, such as with the PUF test (see Section E-4.4). This is especially true when

operating under low-moisture-content conditions where the volume of solution available per cubic meter for collection of samples is small.

Lysimeter experiments should be used as a tool to confirm the coupled glass corrosion and contaminant transport model described in Section E-3.0. For this purpose, there is no need to exclusively test glasses similar in composition to the vendor's formulation or to simulate natural conditions as much as possible. Instead, some experiments should be run with an applied steady flow rate of water and with a glass designed to have high corrosion rate so that it is easier to monitor contaminant release and transport during the test. It is also important that these experiments be designed and instrumented appropriately so that data can be collected to support validation studies of other key models being used on the ILAW disposal program, such as a multiphase flow simulator.

E-6.0 MODELING THE DISPOSAL SYSTEM

The previous discussion of the testing strategy would not be complete without a discussion of the link to modeling the disposal system, as this is the ultimate use of the data generated by the testing program. The fundamental objective of the performance assessment for disposing of ILAW is to calculate the radiation dose to a future population as a result of any release and transport of radionuclides to the unconfined aquifer located approximately 70 m below the disposal facility. Computer models will be used to simulate the processes controlling the release and transport of radionuclides to the unconfined aquifer. The computer codes must perform three major simulation functions: 1) release of contaminants from the vitrified waste, 2) transport of those contaminants through the engineered system, and 3) transport through the vadose zone to the groundwater. In Section E-3.2, we presented the kinetic rate law for glass corrosion:

$$J_i^a = \nu_i k e^{-\frac{E_a}{RT}} a_{H^+}^{-\eta} \left[1 - \left(\frac{Q}{K} \right)^{\sigma} \right], \qquad i = 1,2,...N$$

Assuming that k, E_a, η, K, and σ are all known parameters developed from the laboratory testing program, to determine the mass flux of any component i released from the glass to the aqueous phase, it is necessary to calculate the pH and ion activity product, Q. Because the calculation of these chemical variables depends on both the physical properties of the system, such as flow rate and glass surface area, and chemical properties, such as solubility products and the amounts and types of alteration products formed, functions (1) and (2) discussed above cannot be decoupled. A special type of computational model, called a reactive chemical transport model, is required for simulations.

In 1995, the Hanford Low-Activity Waste Disposal Project selected a reactive transport code to calculate contaminant release rates from the engineered components of the disposal system (McGrail and Mahoney 1995). The **A**nalyzer of **R**adionuclid**E** Source **T**erm with **C**hemical **T**ransport (AREST-CT) code describes multicomponent reactive transport in an isothermal, partially saturated, porous medium. The model includes chemical reactions between aqueous, gaseous, and solid phases. Reactions involving minerals are described through appropriate kinetic rate laws, along with a special option for treating irreversible reactions (such as glass corrosion) via the kinetic rate, Equation (G.2). Homogeneous reactions within the aqueous phase are assumed to be reversible with their reaction rates controlled by transport and local equilibrium mass action relations. Local equilibrium between a gas or gas mixture and the aqueous phase is treated through Henry's Law. Solute transport includes contributions from advection, diffusion, dispersion, and radioactive decay.

Because of evidence from PUF experiments that significant changes in hydraulic properties may occur as a result of precipitation of secondary phases, it was decided in 1997 to incorporate a multiphase flow and transport capability into AREST-CT so that these property

changes could be coupled to the flow field. The specific mechanisms incorporated in the code are changes in porosity, and hence hydraulic conductivity, caused by mineral precipitation-dissolution and changes in water saturation caused by water consumption-production in chemical reactions. This was accomplished by coupling AREST-CT with STOMP, a nonisothermal, multiphase flow simulator (White and Oostrom 1996). The new coupled code is called STORM, **S**ubsurface **T**ransport **O**ver **R**eactive **M**ultiphases.

As it is currently configured, the STORM code represents a relatively complete model of the physical and reactive chemical transport processes that are required for simulating radionuclide release rates from the disposal system. However, several improvements to the code are needed to support long-term performance calculations, especially in more than one spatial dimension. These improvements can be classified in terms of 1) subprocess models and 2) numerical methods.

E-6.1 SUBPROCESS MODEL DEVELOPMENT

STORM has two subprocess models requiring further development: 1) adaptive reaction network and 2) composition-dependent hydraulic property model.

E-6.1.1 Adaptive Reaction Network

The set of reactions to be considered in a single STORM run, the reaction network, is fixed as specified in the input file. Consequently, careful consideration must be given to including all important solid and solution species that may be important in the system. This is extremely difficult because the system being modeled is usually changing chemically in time and space, often in ways that cannot be predicted a priori. Consequently, solid and solution species that were important at the start of a run can become unimportant components in the reaction network and vice versa. A subprocess model is needed in STORM that adapts the reaction network periodically. This can be accomplished by conducting a phase boundary search at each node. The phase boundary search is a numerical implementation of the mineralogic phase rule. It operates by querying a general thermodynamic database and computing an ion activity product for each solid that could exist in the current system based on the elements being considered. Supersaturated phases are added to the reaction network, and undersaturated phases can be deleted. However, because the dissolution rate of a phase is finite, deletion of the phase from the network is constrained until the total mass falls below some specified value.

E-6.1.2 Composition-Dependent Hydraulic Property Model

Unsaturated flow experiments with glass waste forms have shown that the dissolution of the glass and subsequent formation of secondary minerals, such as zeolites and clays, can change the unsaturated flow properties of the glass (McGrail et al. 1997c.). Constitutive equations are

used in computational modeling of multiphase flow to relate changes in flow properties, such as hydraulic conductivity and permeability, to changes in primary variables, such as matric potential or volumetric water content (Tietje and Tapkenhinrichs 1993). For example, a water-retention function developed by van Genuchten (1980) is commonly used in modeling unsaturated flow in porous media

$$\theta = \theta_r + (\theta_s - \theta_r)/[1+(\alpha\psi)^n]^m$$

where θ, θ_r, and θ_s are water content, residual water content, and saturated water content, respectively, ψ is matric potential, and α, n, and m are fitted parameters that are related to the physical properties of the porous medium. Physical properties, such as porosity, tortuosity, air entry matric potential, residual moisture content, etc., are typically assumed as invariant properties of the matrix, and so α, n, and m are assumed to be constant as well. Such assumptions are invalid in chemically reactive systems where macroscopic changes in physical and chemical properties can be induced by interphase mass transfer reactions that affect the pore scale. In general, the most important mass-transfer reactions involve dissolution of primary solid phases and concomitant precipitation of secondary solid phases.

A fundamental physicochemical basis for modifying the empirical fitting coefficients used in constitutive relationships, like the van Genuchten function, for multiphase flow through a system containing fractured glass should be developed. One possible approach to develop an appropriate model is to conduct experiments where hydraulic and physicochemical property data are obtained on uncorroded and corroded glass samples, possibly obtained from PCT and PUF experiments. This data set will form the basis for developing physicochemical constitutive relationships for unsaturated flow. These relationships can then be tested by comparing computer simulations with measurements of the solid phase and moisture distribution during PUF tests (see Section E-4.4).

E-6.2 NUMERICAL METHODS

The vitrified waste may be in the form of large glass blocks riddled with stress fractures. Berkowitz, Bear, and Braester (1988) suggested that solute transport in fractured media can be considered at a number of different scales. A near-field scale would include a few discrete fractures near the source. At a far-field scale, the fractured media could be treated as a continuum that is representative of an equivalent porous medium in which the repeating fractures behave as large pores. Their work focused on a contaminant source surrounded by a fractured porous medium. The fractured glass waste packages emplaced in a vault (Figure E-2.1) presents a different situation; the contaminants are already distributed within the glass matrix and are released as the glass dissolves. Given the low flow rates through the vault, the contaminants will be mixed by diffusion within each fracture. If stress fractures in the glass waste form are numerous and closely spaced, a continuum approach to modeling flow and transport through the vault will likely be adequate. Consequently, developing a model capable of handling flow and transport in discrete fractures is not warranted at this time.

Currently, STORM executes on a fixed, two-dimensional cartesian finite grid. This makes the code inefficient when handling sharp concentration fronts that commonly develop in reactive transport simulations. Consequently, implementing an adaptive gridding algorithm is recommended. Adaptive gridding provides a means to increase the number of computational cells in regions where sharp concentration fronts exist and coarsens the grid where concentrations are uniform. This refinement better captures the movement of these fronts with less numerical dispersion and increases execution speed by putting extra grids at only those locations where they are needed.

Two-dimensional simulations of coupled unsaturated flow and reactive transport with STORM require significant computing resources. A two-dimensional simulation with 22 aqueous species, 9 solid species, 10 equilibrium reactions, and 10 kinetic reactions on a 30 by 35 grid requires 1 week to reach a simulation time of 20,000 years running on a Sun Ultra 1. A sensitivity analysis consisting of multiple two-dimensional runs would therefore take years to complete.

Several alternatives exist for decreasing the execution time of the code. The simplest is to execute the simulations on a workstation with a faster scalar central processing unit (CPU). Workstations are available that are several times faster than a Sun Ultra 1; within 5 years, workstations will be available that are 10 to 100 times faster than a Sun Ultra 1. A second alternative is to adapt STORM for execution on a machine with multiple, parallel processors. Currently, an effort is underway at PNNL to develop a parallel multiphase flow and reactive transport code, targeted for execution on an IBM NWMPP1 computer, capable of 247 GFLOPS. However, the reactive transport algorithm used in this code, operator-splitting, is likely to be restricted to much smaller time steps than the global-implicit algorithm used in STORM. Once this new code is available, testing will be performed to determine the relative efficiency of two codes in running an actual ILAW disposal system simulation. If significant improvements in execution time are demonstrated on a realistic simulation, then the techniques and algorithms developed from the research program on parallel multiphase flow and reactive transport codes may be implemented in a parallel version of the STORM code so that it may be used to solve the larger problems required for the ILAW disposal system performance assessment.

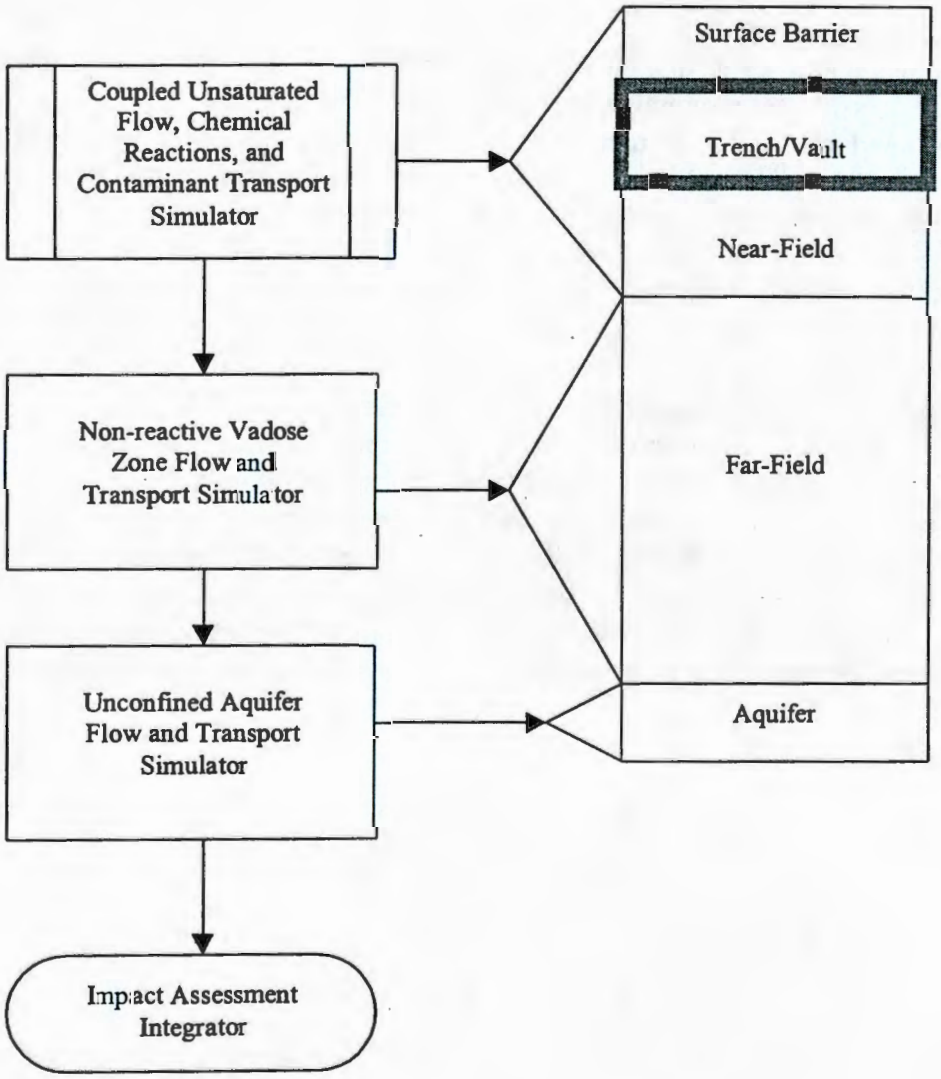
E-6.3 DISPOSAL SYSTEM SIMULATIONS FOR THE PERFORMANCE ASSESSMENT

The ultimate objective of the laboratory and field testing, and model development activities discussed in the previous sections of this report is to provide technically credible calculations of radionuclide release and transport in support of a formal performance assessment for the ILAW disposal system. The general methodology (Mann et al. 1998) is to divide the problem into logical parts that correspond to the computer simulation tools that will be applied in different parts of the problem domain. Figure E-6.1 illustrates the recommended overall computational strategy. The very-near-surface infiltration rate provides a key boundary condition for the remainder of the simulations. The coupled unsaturated flow, chemical reactions, and

contaminant transport simulator (see Section E-6.0) is applied from just below the root zone to some distance into the soil (probably several meters) below the floor of the disposal vault. This region is defined as the near field. Water exiting the region near the vault is expected to be of high ionic strength and pH and this plume will migrate down into the soil column for some distance until dispersion and chemical interactions with the soil components attenuate it. Beyond this depth, the chemical composition of the migrating fluid will likely change very little. Consequently, it is possible to limit the domain over which computationally intensive reactive chemical transport simulations must be performed by switching to a relatively simple vadose zone flow and transport simulation in the far-field domain. The radionuclide flux exiting the vadose zone to the unconfined aquifer is computed with this model and is used as a boundary condition for the unconfined aquifer flow and transport simulator. The final step in the methodology is to compute the impacts, if any, from ingestion, inhalation, and external radiation to humans exposed to the contaminants by withdrawing water from the aquifer and using it for drinking, farming, and other purposes.

The methodology outlined in Figure E-6.1 is remarkably robust for conducting a performance assessment. Each simulation tool is based on basic principles of physics, chemistry, and thermodynamics. No ad hoc assumptions are made about the performance of the waste form or other components of the engineered and natural system. Consequently, changes in boundary conditions, such as infiltration rate, or scenarios, such as an assumed failure of the capillary break, can be quantitatively assessed in terms of their overall impacts on system performance. Of course, the database needed to support the mechanistic models used in the selected simulation tools is substantial. In this report, we have attempted to define the minimum required data set to support the coupled unsaturated flow, chemical reactions, and contaminant transport simulator (see Section E-4.0). Although a considerable amount of testing is required, it is not intractable. Similar statements can be made about the other parts of the methodology. Consequently, the approach outlined in Figure E-6.1, and especially the laboratory and field testing required to support it, are not unreasonable or unrealistic, assuming current programmatic funding levels and schedule are maintained.

Figure E-6.1 Modeling Strategy for Assessing ILAW Disposal System.



E-7.0 CONCLUSION

An overall strategy for evaluating the long-term performance of a low-activity waste glass at the Hanford Site has been presented. The strategy combines laboratory testing and field testing into an overall plan for demonstrating a scientific understanding of the processes controlling long-term glass corrosion. Models describing these processes can then be used with confidence in extrapolating to the disposal system conditions and calculating radionuclide release rates in a formal performance assessment.

The recommended laboratory testing includes single-pass flow-through (SPFT), product consistency (PCT), and pressurized unsaturated flow (PUF) tests. The majority of the laboratory testing (~80%) is to be conducted using nonradioactive glasses, with the remainder performed with glasses containing a selected set of key radionuclides. Additionally, a series of PUF experiments with a natural analog of basaltic glass are recommended to confirm that the alteration products observed under accelerated conditions in the PUF tests are similar to those found associated with the natural analog. This will provide additional confidence in using the PUF test results to infer long-term corrosion behavior.

The final component of the strategy is a set of field experiments using both vendor glasses and specially formulated highly reactive glasses. The purpose of these experiments is to validate the models for glass corrosion and reactive chemical transport that form the technical basis for calculating radionuclide release rates in the disposal system. These experiments can be performed in existing lysimeters at the Hanford Site, or in new lysimeters that have been equipped with the latest in monitoring equipment and located near the proposed disposal site. The field tests will be closely coordinated with other tasks in the ILAW performance assessment activity, so that may serve to validate other key model inputs to the performance assessment.

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Appendix F

Quality Assurance

F. QUALITY ASSURANCE

F.1 OVERVIEW

This section describes the quality assurance activities associated with the performance assessment activity. The quality assurance activities involve reviewing the Performance Assessment by recognized subject matter experts and knowledgeable stakeholders. Three separate areas are covered: experimental data collection, computer code use, and analyses. The following sections are arranged according to which organization performs the activity being discussed.

F.2 CH2M HILL HANFORD GROUP, INC.

The River Protection Project (RPP) Immobilized Waste Program quality assurance activities are covered by the RPP Quality Assurance Program Description (CHG 2000) and associated implementing procedures. This program addresses the requirements of the *Quality Assurance Program Description* of the Tank Farms Contractor, which is based on Title 10 *Code of Federal Regulations* (CFR) Part 830.120 and U.S. Department of Energy (DOE) Order 5700.6C (DOE 1991).

CH2M HILL Hanford Group, Inc. (CHG) is responsible for the quality aspects of all work discussed in this performance assessment.

F.3 PACIFIC NORTHWEST NATIONAL LABORATORY

Pacific Northwest National Laboratory personnel conducted their work for the performance assessment according to the appropriate portions of the laboratory's quality assurance (QA) program. This program conforms to 10 CFR 830.120, "Quality Assurance" through implementation of its Subject Based Management System (SBMS). The SBMS (http://sbms.pnl.gov) includes a set of administrative procedures that define how the requirements of 10 CFR 830.120 are implemented. The administrative procedures define controls, policies, and established methods for managing and conducting all aspects of work that affect quality.

A key aspect of the QA program involves technical and peer reviews of procedures, test plans, data, calculations, and test results. The reviews range from verifying that calculations or data reduction have been performed correctly to evaluating the test methodology described in a proposed test plan.

Records generated from all activities are indexed and managed according to QA program requirements. Record-holding facilities are used for long-term records retention and storage.

F.4 OTHER CONTRACTORS

Other organizations, such as Argonne National Laboratory and Fluor Federal Services worked on this activity. Their work was performed under either the quality assurance plans of CHG or the Pacific Northwest National Laboratory.

F.5 HANFORD ENVIRONMENTAL DOSE OVERVIEW PANEL

The DOE, Richland Operations Office, established the Hanford Environmental Dose Oversight Panel (HEDOP) (Schreckhise 1993) to perform the following:

- Ensure that appropriate radiological and nonradiological environmental and health dose assessment methods are used at the Hanford Site
- Ensure that all Hanford Site-related environmental and health dose assessments are technically consistent
- Foster communications among Hanford Site contractors regarding environmental and health dose assessments.

All dose calculations used in this performance assessment have been reviewed and approved for publication by a HEDOP reviewer (Rhoads 1999).

F.6 TECHNICAL REVIEWS OF SUPPORTING DOCUMENTS

External technical reviews by outside experts are being held on specialized topics because of the large amount of technical data used in performance assessments. These reviews are done to ensure that the proper methods, techniques, and resources are used in obtaining the data.

Hanford Site experts reviewed each data package. In addition, experts from outside the Hanford Site reviewed the waste form and geotechnical data packages. The list of external reviewers is given in Table F-1.

F.7 REVIEW OF THE PERFORMANCE ASSESSMENT

DOE's Low-Level Waste Disposal Facility Federal Review Group (LFRG) (DOE 1999c) formally reviewed the 1998 ILAW Performance Assessment (Mann 1998a). Based on this review, the DOE issued the Disposal Authorization Statement (DOE 1999d) accepting the ILAW Performance Assessment . This acceptance is contingent on the following actions being accomplished:

- Providing the LFRG with documentation of the near-term glass test results to provide confidence that the glass performance assumed in the performance assessment can actually be achieved

Table F-1. Data Package External Reviewers.

Area	Reviewer	Affiliation
Geology	Dr. Ann Tallman	Independent (formally Smith College and Westinghouse Hanford Company)
	Newell P. Campbell	Independent (formally Professor of Geology, Yakima Valley College).
Recharge	John Robert Nimmo	United States Geological Survey, Menlo Park, CA
	Bridget R. Scanlon	University of Texas at Austin
Near-Field Hydrology	John Robert Nimmo	United States Geological Survey, Menlo Park, CA
	Bridget R. Scanlon	University of Texas at Austin
Far-Field Hydrology	Lynn W. Gelhar	Civil & Environmental Engineering, Massachusetts Institute of Technology
Geochemistry	Dr. Tjalle (Chuck) Vandergraaf	Atomic Energy of Canada Limited, Whiteshell Research Laboratory in Pinawa, Manitoba
	Dr. Steve Serkiz	Savannah River Technology Center
	Dr. Patrick Brady	Department of Geochemistry, Sandia National Laboratory

- Addressing the secondary issues identified by the review team in future revisions to the performance assessment.

Documentation on relevant glass performance has been provided to the LFRG for their review (French 1999 and French 2000a) and the LFRG has determined that the condition has been met (DOE 2000). The secondary issues identified by the LFRG are addressed in this version of the ILAW Performance Assessment (see Appendix A).

The 1998 ILAW Performance Assessment also was reviewed by the Washington State Department of Ecology (Appendix F of Mann 1998a) and by staff of the U.S. Nuclear Regulatory Commission.

This document was reviewed by the immobilized low-activity waste performance assessment team, as well as by program management of the Immobilized Waste Storage and Disposal Program (K.C. Burgard) and of DOE's Office of River Protection (N.R. Brown, P.E. LaMont, and others). In addition, the lead authors of other Hanford Site performance assessments (C.T. Kincaid [Grout Performance Assessment and Hanford Site Composite Analysis] and M.I. Wood [200 East Area Solid Waste Performance Assessment and 200 West Area Solid Waste Performance Assessment]) performed an overall technical review of this report.

Comments on the draft version of this report by the review team of the Low-Level Waste Disposal Facility Federal Review Group have been incorporated.

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