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Station #12

ENGINEERING DATA TRANSMITTAL

Page 1 of 1

1. EDT 600850

2. To: (Receiving Organization) Distribution	3. From: (Originating Organization) Environmental Engineering Group	4. Related EDT No.: N/A
5. Proj./Prog./Dept./Div.: Environmental Restoration	6. Cog. Engr.: F.V. Roeck	7. Purchase Order No.: N/A
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(A) Item No.	(B) Document/Drawing No.	(C) Sheet No.	(D) Rev. No.	(E) Title or Description of Data Transmitted	(F) Impact Level	(G) Reason for Transmittal	(H) Originator Disposition	(I) Receiver Disposition
1	WHC-SD-EN-TI-201		0	Screening Performance Assessment/Risk Assessment for the Proposed Environmental Restoration Disposal Facility (ERDF)	4	1/2	1	

16. KEY

Impact Level (F)	Reason for Transmittal (G)		Disposition (H) & (I)	
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2	1	Cog.Eng.F.V. Roeck	F.V. Roeck	10/27/93	H6-01	IRA Clearance (2)			H4-24	3	
2	1	Cog. Mgr.M.J. Lauterbach	M.J. Lauterbach		H6-01						
		QA									
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		Env.									
3		Central Files (2)			L8-04						
3		EPIC (~~2~~) 1			H6-08						

18. F.V. Roeck F.V. Roeck 10/27/93 Signature of EDT Originator Date	19. Authorized Representative Date for Receiving Organization	20. M.J. Lauterbach M.J. Lauterbach 10/26/93 Cognizant/Project Engineer's Manager Date	21. DOE APPROVAL (if required) Ltr. No. [] Approved [] Approved w/comments [] Disapproved w/comments

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WHC-SD-EN-TI-201, Rev. 0

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Part 1

SUPPORTING DOCUMENT

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WHC-SD-EN-TI-201

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7. Abstract

WHC, 1993, *Screening Performance Assessment/Risk Assessment for the Proposed Environmental Restoration Disposal Facility (ERDF)*, WHC-SD-EN-TI-201, Rev. 0, prepared by Golder Associates, Inc., and A. T. Kearney, Inc., for Westinghouse Hanford Company, Richland, Washington.

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9. Impact Level 4

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EXECUTIVE SUMMARY

The Environmental Restoration Disposal Facility (ERDF) is intended to receive waste from remediation of waste management operable units within the Hanford Site. This performance assessment/risk assessment screening study was conducted to assist site characterization and preliminary design efforts. Sixteen different alternatives were simulated in this study. The elements that varied between the scenarios included the waste type, the treatment approach, the liner, and the barrier. This study was intended to be a scoping study that relied upon simple analytical tools and readily available information regarding governing processes and parameters. A spreadsheet model was created that relied upon simple analytical expressions for description of contaminant fate and transport. In general, these expressions assume steady-state flow conditions and one-dimensional plug flow.

Performance was measured in terms of maximum risk and travel time at three points of compliance: 1) C_1: the bottom of the trench, 2) C_2: the water table directly beneath the ERDF, and 3) C_3: the saturated zone intercepted by a vertical plane extending down from the boundary of the ERDF. The exposure assessment assumed residential consumption of groundwater at each point of compliance. Other exposure pathways were not addressed. Risk calculations were performed using the approach described in the Hanford Site Baseline Risk Assessment Methodology (HSBRAM) (DOE-RL 1993). With the exception of hexavalent chromium [Cr(VI)], all the simulated contaminants are carcinogens and their health effects can be expressed in terms of an incremental cancer risk (ICR). Because Cr(VI) is a chemical toxin but not a carcinogen (via ingestion), it was necessary to express Cr(VI) health effects in terms of a hazard quotient (HQ). Although it was assumed that Cr(VI) would be a significant contaminant of concern and that Cr(VI) would not be reduced to Cr(III), the presence of Cr(VI) in 100 Area and 300 Area soils has not been verified.

For the compliance point directly beneath the trench, it was necessary to assume that vadose zone pore water is collected and consumed by the exposed individual. Although this pathway is not realistic, it provides a performance measure in relation to the "no-release" criteria. Even the other two compliance points are unrealistic since no wells are located near the proposed facility and it is unlikely that water supply wells will be completed beneath or near the facility as long as the 200 Area remains under institutional control. Given these unrealistic exposure pathways, the risk results presented in this study should be considered performance measures against different criteria, not actual risk estimates. This point is further emphasized by the fact that not all exposure pathways were considered, only six contaminants were addressed, and actual concentrations of those six contaminants will likely differ from those assumed in the simulations.

Both deterministic and probabilistic simulations were performed. The results of the probabilistic Monte Carlo simulations suggest the following conclusions regarding estimated risks:

- Estimated maximum risks directly beneath the trench (at C_1) are high for all the alternatives, with median ICRs ranging from 10^{-4} to greater than 1, and median HQs ranging from 10 to 100.

- Strontium-90, trichloroethylene (TCE), and uranium-238 were generally the prime cancer-causing risk drivers beneath the trench.
- Median ICRs at the ERDF boundary for all the alternatives are in the range of 10^{-5} to 10^{-3}, and are expected to occur between 10,000 and 500,000 years in the future. Uranium is the only cancer-causing contaminant that reached groundwater at significant risk levels.
- Assuming that Cr(VI) is a contaminant, median HQs at the ERDF boundary are in the range of 0.5 to 10 for all the alternatives, and are expected to occur between 500 and 10,000 years in the future.
- None of the cancer-causing constituents were predicted to reach groundwater within 10,000 years for the barrier alternatives; therefore, it is highly likely that the ICR in groundwater for the barrier alternatives will be negligible for the next 10,000 years. For the alternatives without barriers, the probability that uranium would reach groundwater within 10,000 years was approximately 50 percent; in realizations where U-238 reached groundwater the ICR exceeded 10^{-4}.

The results suggested the following conclusions regarding the relative performance of the various configuration elements:

- Because leachate concentrations were limited by solubility or sorption controls, fixation did not result in any reduction in risk compared with untreated waste.
- An infiltration barrier is probably the most critical element of the facility performance.
- Differences between the performance of the modified RCRA barrier and the Hanford barrier were virtually indistinguishable.
- Due to decay during migration of contaminants through the liner, the presence of a liner reduced the ICR at the base of the trench but did not change the HQ (chromium(VI) was assumed not to decay). However, liners did not significantly change the ICRs in the groundwater.
- At C_1, vitrification reduced the ICR for E waste and reduced both the ICR and the HQ for C waste. At C_2 and C_3, vitrification only reduced the HQ for C waste alternatives.

The parameter sensitivity analyses indicated that U-238 solubility, barrier infiltration rate, saturated zone hydraulic conductivity, saturated zone hydraulic gradient, and the vadose zone mixing factor are the most important parameters for determining maximum risk. For alternatives that include vitrification, the dissolution rate of vitrified material and the U-238 partition coefficient are also important. In general, the list of important parameters is reduced when considering relative performance of the alternatives.

Finally, general conclusions resulting from this study include the following:

- The presence of Cr(VI) as a significant contaminant in 100 Area and 300 Area soils should be determined.
- The performance criteria (including time and place of compliance and allowable risk) should be determined as soon as possible to help focus future performance assessments and conceptual design of the ERDF.
- Although the benefits of fixation and vitrification are not warranted by the increase in cost, other types of treatment that result in chemical changes affecting contaminant mobility or solubility may deserve further attention. Because the effectiveness of these types of treatment are dependent on waste characteristics, they should be evaluated in the operable unit RI/FS process.

CONTENTS

CONTENTS (Cont.)

FIGURES:

CONTENTS (Cont.)

FIGURES (cont):

TABLES:

CONTENTS (Cont.)

TABLES (cont):

APPENDICES:

1.0 INTRODUCTION

The Environmental Restoration Disposal Facility (ERDF) is intended to receive waste resulting from remediation of waste management operable units within the Hanford Site (Figure 1-1). For the most part, these operable units will be located along the Columbia River, either in the 100 Area or the 300 Area, although some waste may originate from 200 Area operable units. The primary waste generating activity in the 300 Area was fuel element fabrication, although some waste was generated due to a variety of research and development activities. The primary waste generating activity in the 100 Area was operation of nuclear reactors for the production of plutonium. The majority of the waste in the 100 and 300 Areas is low-level radioactive soil contaminated with uranium and/or fission products. The majority of the waste has sufficiently low concentrations of radionuclides to be considered low-activity, although a small percentage is considered high-activity waste. Other wastes include solid waste from landfills (which are generally poorly characterized) and demolition debris. Hazardous constituents, including metals and organic solvents, are found in some of the wastes.

The proposed location of the ERDF, shown on Figure 1-2, is south of the area between the 200 East and 200 West Areas. The dimensions of the facility are approximately 3,000 m (9,900 ft) by 3,300 m (11,000 ft). It is anticipated that approximately 23 million cubic meters (30 million cubic yards) could ultimately be disposed of in the ERDF.

1.1 PURPOSE

Remediation of the 100 and 300 Areas, which is anticipated to include large-scale excavation and disposal of wastes, cannot substantially begin until the ERDF is constructed. Therefore, design and permitting of the ERDF is a critical element of the overall Hanford Site remediation effort. The primary objectives of this study were to:

- Provide performance information on potential ERDF facility designs and treatment technologies. This information will help focus ERDF design discussions between the regulators and the Department of Energy (DOE) on the most critical issues.
- Identify the most important data needs for further performance and risk assessment of the ERDF.

Both of these objectives were intended to accelerate the conceptual design process. This work is intended as a preliminary scoping study that will provide a starting point for more rigorous performance assessment modeling of the ERDF.

1.2 SIMULATED SCENARIOS

As shown in Table 1-1, sixteen different design and treatment alternatives were simulated in this study. These sixteen alternatives were intended to represent the range of possible alternatives, from no-action (Alternative 1) to highly conservative (Alternative 14), while illustrating the performance of likely alternatives. The elements that varied between the scenarios included the waste type, the treatment approach, the liner, and the surface barrier. Each of these elements are briefly discussed below.

1.2.1 Waste Type

A variety of wastes with a range of contaminants and contaminant concentrations will be disposed of in the ERDF. Performance of the facility will vary depending upon the characteristics of the waste. In order to compare a variety of different designs it was necessary to suppose hypothetical waste types. This study assumed that the waste was contaminated soil; this was a reasonable assumption given that the majority of the waste will be soil. Solid waste (from landfills) and demolition debris were not addressed in this study, due to both their relatively small volumes and the lack of knowledge regarding their physical and chemical properties. Three types of contaminated soil were simulated: high activity mixed waste (Type E), low-activity mixed waste (Type C), and low-activity radioactive waste (Type A). As shown in Table 1-2, the differences between these waste types are manifest as the concentrations of contaminants. These concentrations were generally based upon typical concentrations detected in 100 or 300 Area soils. In some cases, due to lack of detectable concentrations, it was necessary to select a concentration without a suitable analog. The following constituents were modeled:

- Uranium-238 (U-238)
- Strontium-90 (Sr-90)
- Plutonium-239 (Pu-239)
- Hexavalent Chromium [Cr(VI)]
- Trichloroethylene (TCE)
- Polychlorinated biphenyls (PCBs)

Primary criteria for selection of simulated constituents were: 1) prevalence in 100 and 300 Area soils, 2) significantly elevated concentrations detected in groundwater beneath the 100 and 300 Areas, 3) suitability as a surrogate for other constituents, and 4) the desire to represent different categories of contaminants with a broad spectrum of different fate and transport characteristics. As described below, these constituents address a broad range of transport and fate characteristics.

U-238 is an extremely long-lived (half-life of 4.5 billion years) relatively mobile radionuclide. It is a suitable surrogate for the other uranium isotopes. Elevated uranium concentrations are found in soil and groundwater in the 300 Area and the 100-H Area.

Sr-90 is a relatively short-lived (half-life of 30 years) moderately sorbed radionuclide. It is a suitable surrogate for cesium-137 (Cs-137). Sr-90 is found in both soil and groundwater throughout the 100 Area.

Pu-239 is a long-lived, strongly-sorbed radionuclide. It is not found in groundwater but has been detected in soils in the 100 Area.

Cr(VI) is a very mobile metal that does not decay, although it can be reduced to trivalent chromium, which is immobile and much less toxic. Chromium is the most prevalent non-radioactive metal contaminant found in 100 Area soils, although it is uncertain whether it is present as Cr(III) or Cr(VI). The usefulness of simulating Cr(VI) is debatable because it is uncertain whether or not it is present in vadose zone soils. Cr(VI) compounds were added to cooling waters throughout the 100 Area facilities to inhibit corrosion in pipes and was discharged to the subsurface in liquid effluents. Although Cr(VI) has been detected in groundwater, it has not yet been measured in soils at significantly elevated concentrations. Although total chromium has been found in vadose zone soils, it is probably Cr(III). This is because the non-sorbed Cr(VI) would have flushed through the vadose zone in the liquid effluent discharge leaving behind the strongly-sorbed Cr(III). This hypothesis could be investigated by measuring Cr(III) and Cr(VI) in soils that are known to have elevated concentrations of total chromium. Cr(VI) is a suitable surrogate for Tc-99.

TCE is a relatively mobile chlorinated compound that is a suitable surrogate for volatile organic solvents in general. Although rarely found in the 100 or 300 Area soils, spent solvents such as TCE are probable wastes in land disposal facilities. TCE and other chlorinated solvents are not typically considered readily degradable. However, given the slow migration rates expected beneath the ERDF, degradation must be considered.

PCBs are a class of relatively immobile and persistent organic compounds. They have been detected at low concentrations in a few places in the 100 and 300 Area. Similar to TCE, degradation must be considered given the slow migration rates.

1.2.2 Treatment

In addition to scenarios with no treatment, three types of treatment were considered: soil washing, fixation (or grouting), and vitrification.

Soil washing was assumed to concentrate the waste constituents into 20 percent of the original soil volume. Soil washing was not considered for E waste because the concentrations are already high enough that considerable radiation protection measures will be required for any handling of the waste. Concentrating E waste further will only increase the hazards to workers and the need for radiation protection measures.

Fixation is currently used at Hanford for treatment of high-level tank waste and has been proposed for treatment of low-level waste. For this study, fixation was assumed to provide physical containment only. Although additives can be included in the grout to change the chemical behavior of a specific contaminant, such additives were not considered. These additives are an area of ongoing research and data were not located regarding how they affect chemical behavior of the six simulated contaminants.

Vitrification has been proposed for treatment of high-level tank waste at the Hanford Site. Vitrification of all the waste from the 100 and 300 Areas may not be feasible but it was addressed in this study to illustrate the performance aspects of the extremely conservative end of the treatment spectrum.

1.2.3 Liners

In addition to unlined landfill scenarios, three types of liners were considered: a double lined minimum technology requirements (MTR) liner, a single liner, and a concrete vault.

A cross-section of the MTR liner is shown in Figure 1-3. This liner consists of a 60 cm (24 in.) thick soil protection layer over two flexible-membrane liners (along with associated drainage layers), on top of a 0.9 m (3 ft) thick compacted clay layer. The single layer is the same as the MTR liner except it only has one flexible membrane liner. The concrete vault would be on the order of 45 m (150 ft) wide and 7 m (23 ft) deep, with a wall thickness of 0.6 m (2.0 ft) and a bottom thickness of 0.5 m (1.7 ft), and an asphalt coating.

1.2.4 Barriers

In addition to scenarios with no barriers, two types of surface barriers were considered: the Hanford barrier and the modified RCRA barrier.

The Hanford barrier is shown in Figure 1-4. It consists of nine layers with a total thickness of 4.5 m (15 ft). The upper two layers consist of low permeability soils that create a water retention and evaporation zone. The next four layers are primarily coarse-grained materials that create a capillary break and biotic intrusion barrier. The final three layers consist of drainage rock over low permeability asphaltic concrete over base course.

The modified RCRA barrier, shown in Figure 1-5, consists of seven layers and a total thickness of 1.7 m (5.6 ft). The layers are similar to the Hanford barrier but tend to be thinner. For example, the upper two layers are reduced from 100 cm (40 in.) each in the Hanford barrier to 50 cm (20 in.) each in the modified RCRA barrier. Although the name is misleading, this barrier is more similar to the Hanford barrier than a standard RCRA barrier.

1.3 APPROACH

This study was intended to be a scoping study that relied upon simple analytical tools and readily available information regarding governing processes and parameters. Information regarding governing processes and parameters was available to varying degrees. Preference was given to Hanford site-specific information but it was necessary to utilize non-Hanford information in some cases. Although literature searches were conducted for a variety of processes and parameters, these searches were not exhaustive. If deemed necessary, more extensive literature searches may be warranted.

A spreadsheet model was created that relied upon simple analytical expressions for description of contaminant fate and transport. The analytical expressions are explained in Chapter 2. In general, they assume steady-state flow conditions and one-dimensional plug flow. Although multi-dimensional, transient modeling would allow better simulation of residual drainage, dilution, and lateral spread, these effects are likely to be insignificant compared with constituent retardation, solubility, and infiltration. Residual drainage is not a significant factor for modeling of the ERDF because no liquid effluent has been disposed of near the site and the vadose zone moisture conditions will be at steady state. As explained in Chapter 2, transverse spreading and dilution were addressed using a mixing factor approach. Barrier infiltration modeling presented in Appendix A demonstrates that, although precipitation events are highly transient, the moisture flux is buffered in the upper moisture retention layers and the deep percolation rate is virtually unchanging with time.

Performance was measured in terms of maximum risk and travel time at three points of compliance: 1) the bottom of the trench, 2) the water table directly beneath the ERDF, and 3) the saturated zone intercepted by a vertical plane extending down from the boundary of the ERDF. Risk provides a useful performance measure because it allows combining the effects of multiple contaminants into two measures (incremental cancer risk and hazard quotient). The exposure assessment assumed residential consumption of groundwater at each point of compliance. Other exposure pathways were not addressed. Risk calculations were performed using the approach described in the Hanford Site Baseline Risk Assessment Methodology (HSBRAM) (DOE-RL 1993). With the exception of Cr(VI), all the simulated contaminants are carcinogens and their health effects can be expressed in terms of an incremental cancer risk (ICR). Because Cr(VI) is a chemical toxin but not a carcinogen via the ingestion pathway, it was necessary to express Cr(VI) health effects in terms of a hazard quotient (HQ). This is further explained in Section 2.7.

For the compliance point directly beneath the trench, it was necessary to assume that vadose zone pore water is collected and consumed by the exposed individual. Although this pathway is very unlikely, it provides a performance measure against the "no-release" criterion. Even the other two compliance points are unrealistic since no wells are located near the proposed facility and it is unlikely that water supply wells will be completed beneath or near the facility while the 200 Area remains under institutional control. It is important that risk management distinguish between actual risk and hypothetical risks. Therefore, risk results presented in this study should be considered performance measures, not actual risk estimates. This point is further emphasized by the fact that not all exposure pathways were considered, only six contaminants were addressed, and actual concentrations of those six contaminants will differ from those assumed in the simulations.

Deterministic simulations were conducted using best estimates of governing processes and parameters. The deterministic results are presented in Section 4.1. Best estimates were considered more appropriate than conservative estimates because the primary objective was a relative comparison of alternative scenarios, not to determine the viability of a scenario. Uncertainty analyses, presented in Section 4.2, were conducted to determine the likely range of results. Sensitivity analyses, presented in Section 4.3, were conducted to determine which parameters were most important. Cost estimates for each of the alternatives are developed in Appendix C.

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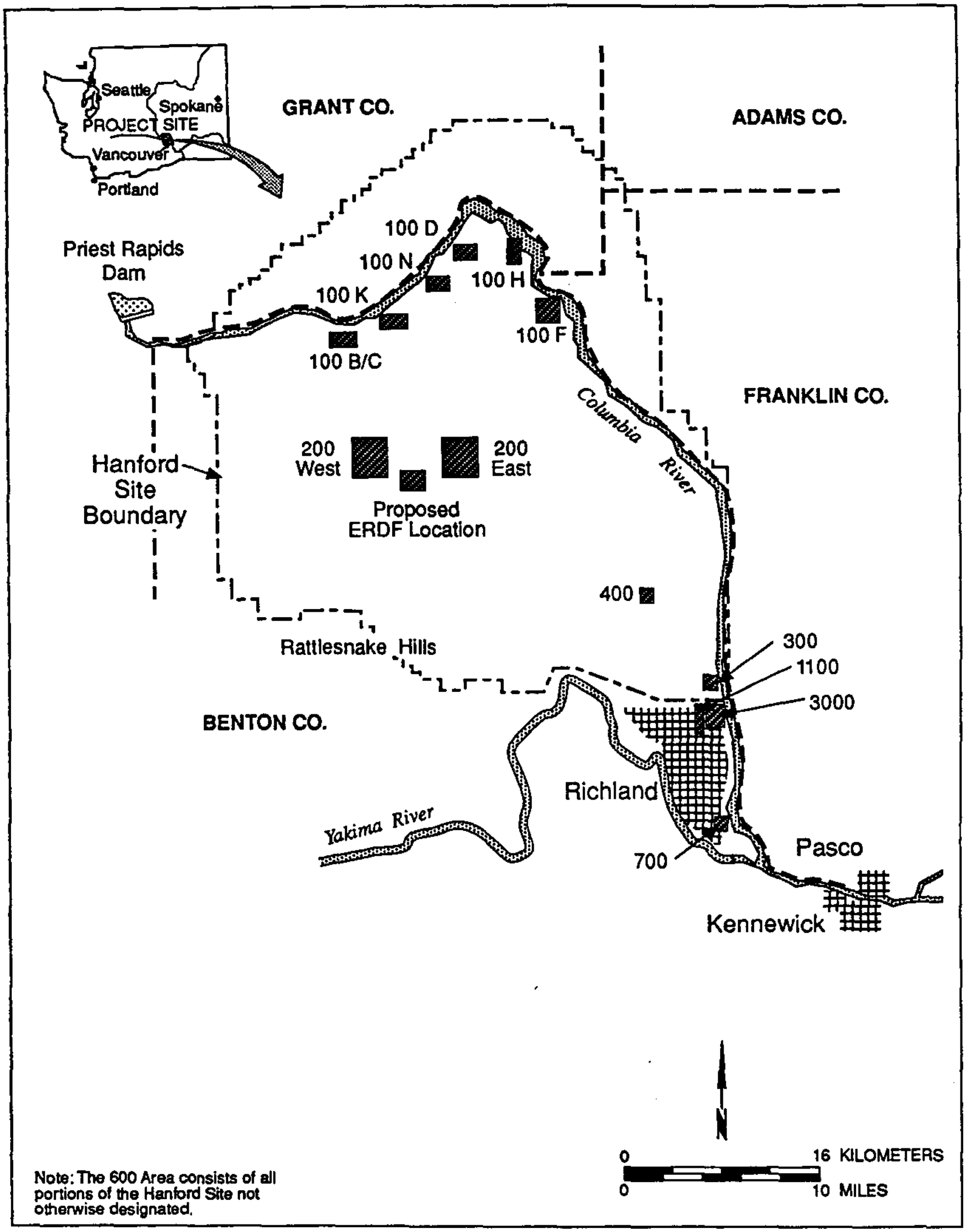

Note: The 600 Area consists of all portions of the Hanford Site not otherwise designated.

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Figure 1-1. Hanford Site Operational Areas.

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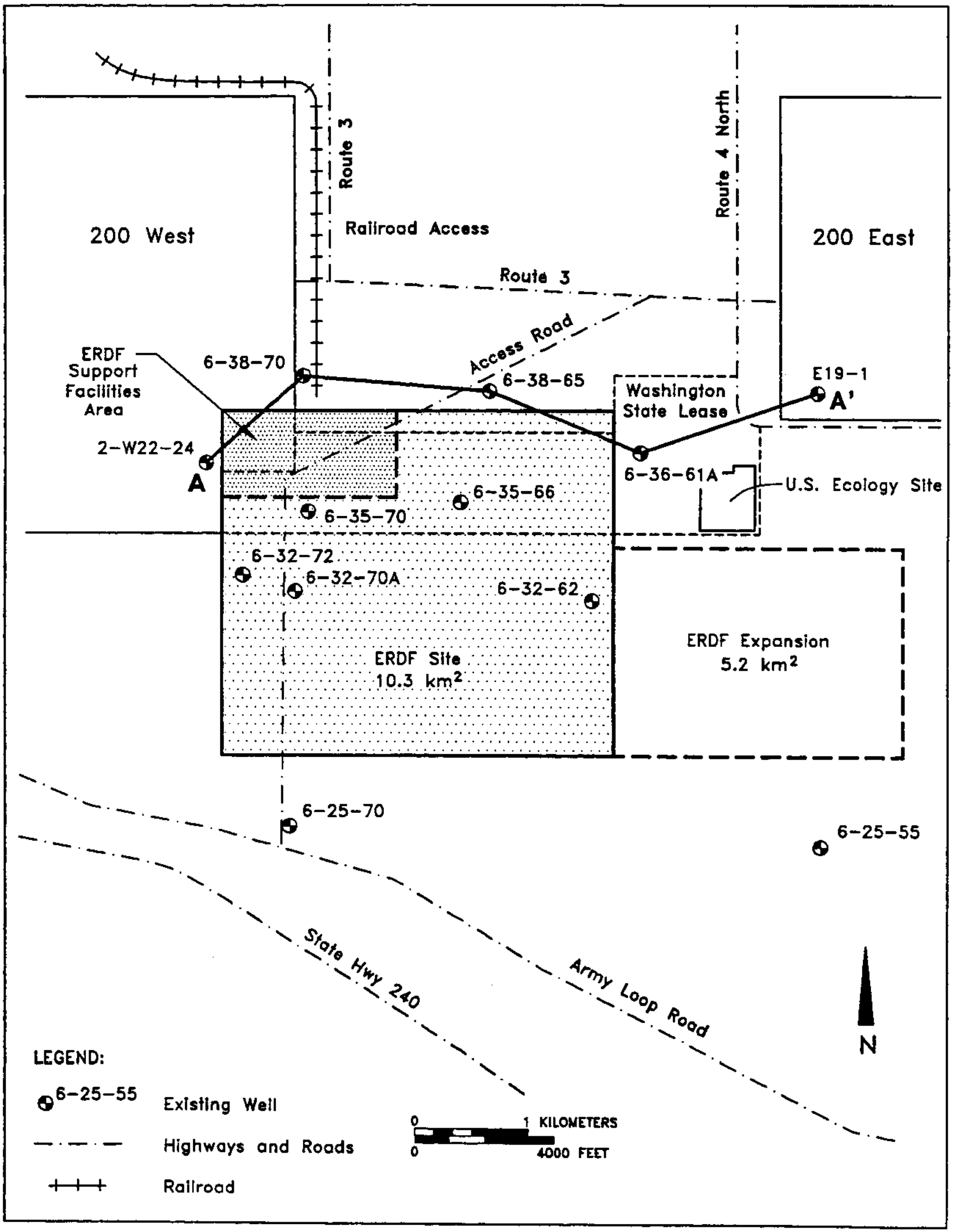

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Figure 1-2. Proposed Location of the ERDF Site.

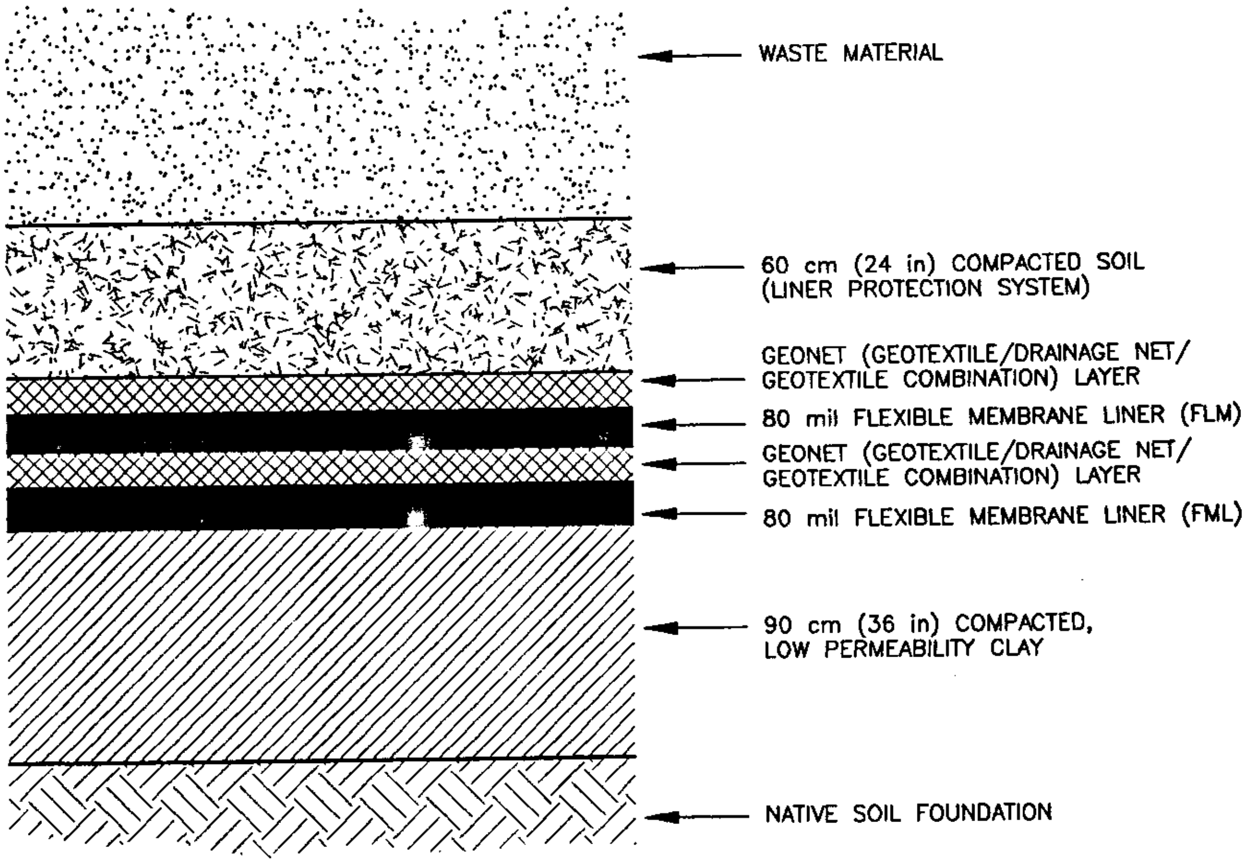

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Figure 1-3. Cross Section of Minimum Technology Requirements (MTR) Liner.

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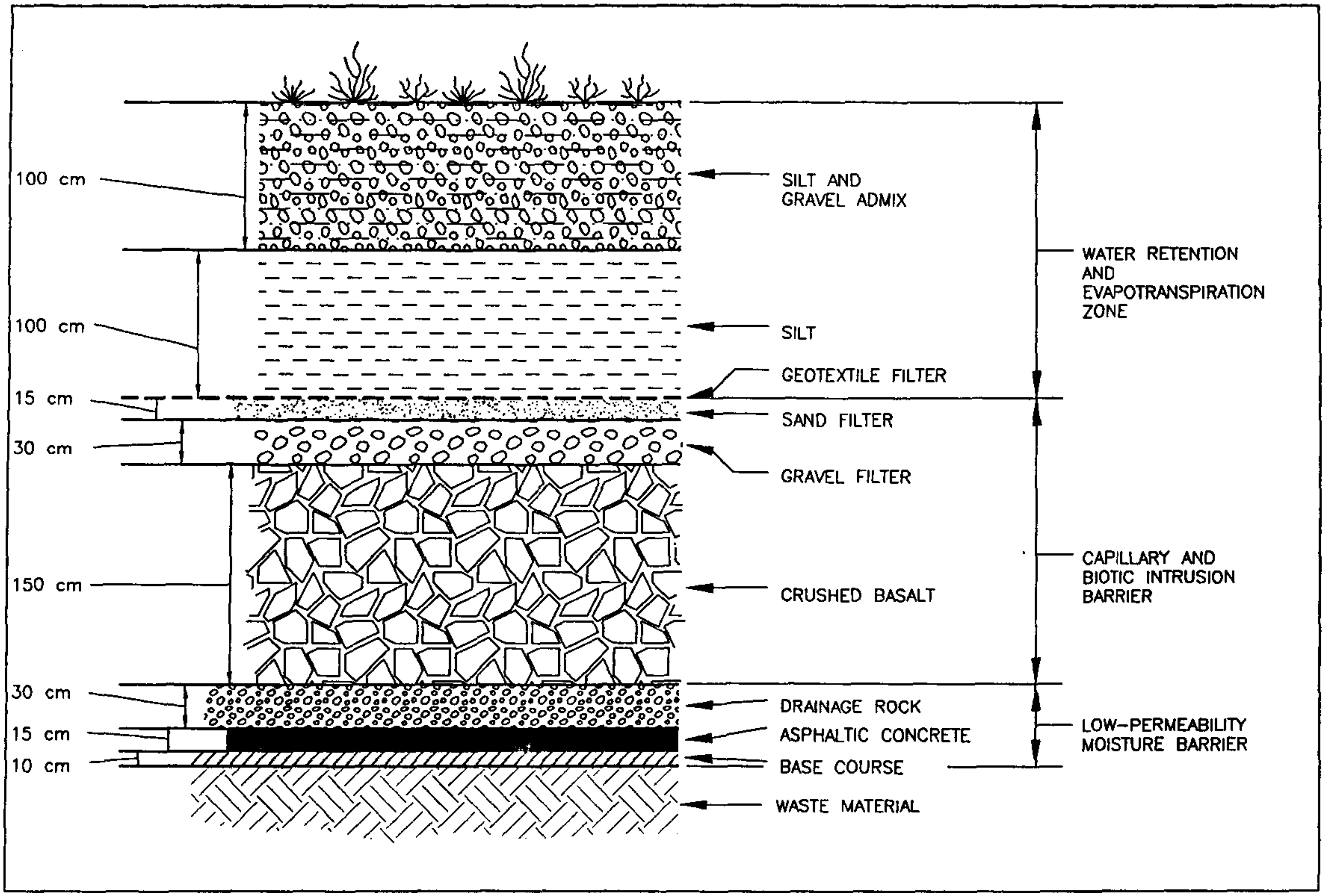

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Figure 1-4. Cross Section of Hanford Barrier.

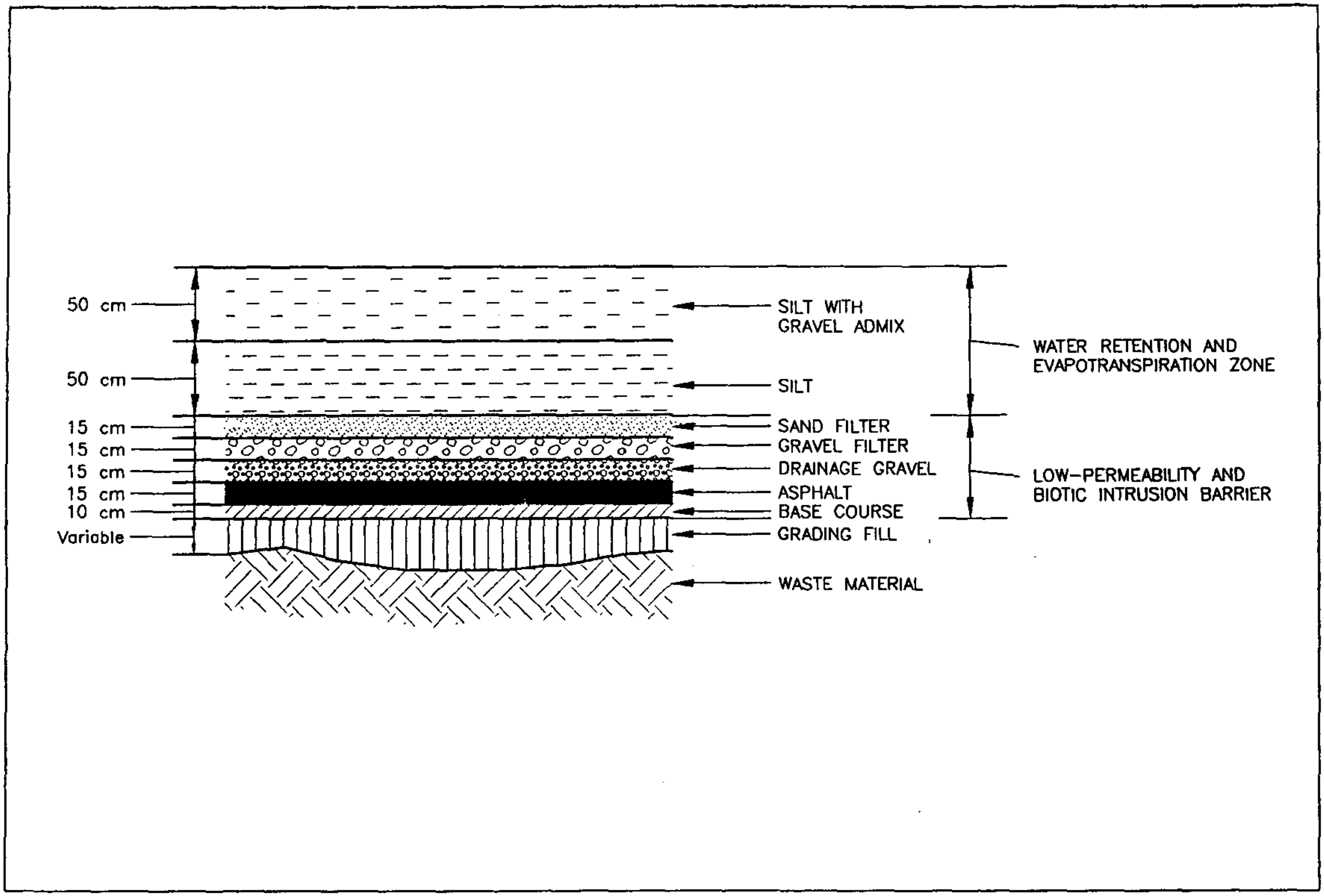

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Figure 1-5. Cross Section of Modified RCRA Barrier.

Table 1-1 Components in Each Alternative

Alternative	Waste type	Soil Washing	Treatment	Liner	Barrier
1	E	No	No Treatment	No Liner	No Barrier
2	E	No	No Treatment	No Liner	Hanford
3	E	No	No Treatment	MTR	RCRA
4	E	No	No Treatment	Single Liner	RCRA
5	E	No	Fixation	MTR	RCRA
6	E	No	Vitrification	No Liner	No Barrier
7	C	No	No Treatment	No Liner	No Barrier
8	C	Yes	No Treatment	No Liner	Hanford
9	C	No	Fixation	MTR	RCRA
10	C	Yes	Vitrification	No Liner	No Barrier
11	C	No	No Treatment	Single Liner	Hanford
12	C	Yes	Fixation	Single Liner	Hanford
13	A	No	No Treatment	No Liner	No Barrier
14	A	Yes	Vitrification	Vault	RCRA
15	C	No	Fixation	No Liner	Hanford
16	C	No	Fixation	No Liner	No Barrier

Note: A-waste: low-activity wastes; C-waste: low-activity, low-concentration mixed (RCRA) wastes; E-waste: high-activity, high-concentration mixed (RCRA) wastes.

Table 1-2. Contaminant Concentrations for Each Waste Type

Waste Type	U-238	Sr-90	Cr (VI)	TCE	PCBs	Pu-239
A Waste Activity Concentration (pCi/gm)	100	100	NA	NA	NA	10
C Waste Activity Concentration (pCi/gm)	100	100	NA	NA	NA	10
E Waste Activity Concentration (pCi/gm)	1.0E+04	1.0E+06	NA	NA	NA	1.0E+03
A Waste Mass Concentration (mg/kg)	300	7.3E-07	0	0	0	0.16
C Waste Mass Concentration (mg/kg)	300	7.3E-07	10	10	0.1	0.16
E Waste Mass Concentration (mg/kg)	3.0E+04	7.3E-03	1.0E+03	100	0.1	16

NA - Not Applicable.

2.0 MODEL FORMULATION

The objective of this study was to evaluate and compare the performance of various trench disposal design alternatives for the proposed ERDF facility. As discussed in the previous chapter, design alternatives differ by waste type, barrier type, liner type, and level of treatment. Performance was measured in terms of maximum risk and travel time to three compliance points: 1) just below the trench liner; 2) at the water table; and 3) at the facility boundary. This chapter describes the analytical approach for calculation of maximum constituent concentrations, constituent travel times, and risks at each compliance point.

2.1 GENERAL APPROACH

The processes controlling the concentrations (and hence the risk) at various points in the system are complex: they are highly coupled, unsteady (i.e., time varying) and, in many cases, non-linear. A detailed performance assessment would involve simulating the system through time using a total system numerical model specifically formulated to incorporate non-linear dynamic processes. Although such models have been developed (Barnard et al. 1992, and Miller et al. 1993) and applied to other sites, such an exercise was beyond the scope and time frame of the present effort. Instead, analytical approximations were used to determine maximum concentrations at various compliance points for each constituent of interest. This approximate approach attempted to consider all major controlling processes, while still remaining analytically tractable. The equations described in the following sections are implemented in a spreadsheet model.

In order to evaluate system performance using the analytical approximations described below, the following major assumptions have been made:

- All input parameters are time invariant (although decay is accounted for).
- Discrete disruptive events (such as earthquakes, volcanic activity, or human intrusion) or gradual deterioration (such as erosion) which may affect the facility are not considered.
- All travel time calculations assume plug flow (i.e., no longitudinal dispersion).
- The synthetic materials in the liners are expected to deteriorate or breach relatively rapidly and are not included in the simulated liners. Consequently, the MTR liner and the single liner were modelled as equivalent.
- Climatic conditions are assumed to remain the same over the duration of the simulations.
- It is assumed that before final closure the facility is operated such that no contaminated leachate is generated. This may be

accomplished by placing the wastes at moisture contents below their retention capacity, or by keeping the wastes covered during operations.

- For cases that include soil washing, it is assumed that the chemical and physical characteristics of the waste are unchanged (other than increasing contaminant concentrations).

Additional assumptions are discussed in subsequent sections.

2.2 COMPUTATIONAL ALGORITHM

As mentioned above, the model presented here estimates the maximum concentration at each of three locations for each constituent. Lifetime incremental cancer risk (ICR) or hazard quotient (an indication of systemic toxic effects) are calculated based on these concentrations.

The four points at which concentrations are computed are shown in Figure 2-1. C_0 is the initial leachate concentrations at the bottom of the waste. C_1 is the maximum leachate concentration at the base of the trench (below the liner, if present); C_2 is the maximum groundwater concentration at the water table (before mixing in the saturated zone); and C_3 is the maximum groundwater concentration in the saturated zone at the facility boundary.

The first step in the calculation is to compute (for each constituent) the maximum leachate concentration above the liner (i.e., the source concentration). This is referred to as C_0. C_1 is then computed directly as a function of C_0; C_2 is computed directly as a function of C_1; and C_3 is computed directly as a function of C_2.

The remaining sections of this chapter discuss in detail the computational model used to carry out the performance calculations. Section 2.3 describes the computation of the maximum source concentration, C_0, within the disposal facility. Section 2.4 discusses transport through the liner and computation of C_1. Section 2.5 discusses transport through the vadose zone and computation of C_2. Section 2.6 discusses transport through the saturated zone and computation of C_3. Exposure assessment and risk characterization (calculated risks based on concentrations) are discussed in Section 2.7.

2.3 SOURCE CONCENTRATION (C_0)

As mentioned above, in order to compute C_1, it is necessary to first compute C_0, the maximum concentration above the liner. Most of the simulations assume that the permeability of the liner is always greater than that of the barrier. This ensures that the trench never acts as a "bathtub", whereby it gradually fills with water. Instead, it is assumed that the rate of seepage through the liner is equal to the rate of infiltration through the cap.

As discussed in Section 1.2.2, treatment can affect the calculation of C_0. Soil washing acts to concentrate the contaminants into a smaller volume of waste; for this study, it was assumed that soil washing increased the concentrations by a factor of five and volume was reduced by a factor of five. The other treatment approaches, fixation and vitrification, can affect the rate of release from the waste. Calculation of C_0 for untreated waste, fixated waste, and vitrified waste, is described below.

2.3.1 Calculation of C_0 for Untreated Waste

For untreated waste, it is assumed that C_0 is controlled by the solubility of the contaminant, the amount of contamination in the waste soil, and the partition coefficient between water and soil for the contaminant. Assuming that the contaminant has reached equilibrium between the soil and the pore water, C_0 can be computed as follows:

$$C_0 = MIN\left[\frac{M_w}{(K_{d,w}+\frac{\theta_w}{\rho_w})}, C_{sol}\right] \qquad (2\text{-}1)$$

where:

M_w = concentration of contaminant in waste (mg/kg);

$K_{d,w}$ = partition coefficient between waste and water (L/kg);

θ_w = volumetric moisture content of the waste (unitless);

ρ_w = dry density of the waste (kg/L)

C_{sol} = solubility of contaminant in water (mg/L).

This equation indicates that the concentration is controlled by the sorption equilibrium, with the constraint that the concentration can never exceed the solubility.

Given these assumptions, C_0 should decrease with time (since M_w decreases due to decay). We are interested in the maximum concentration, and therefore compute C_0 prior to decay (using the initial value of M_w). For radioactive isotopes, the solubility of a particular species (more correctly referred to as the saturation concentration of the species) is equal to the solubility of the element multiplied by the isotopic fractionation (relative abundance) of the species. However, the radionuclides considered in this study are the dominant isotopes and isotopic fractionation is not incorporated into the calculations. This is a reasonable approach given the large uncertainty associated with the radionuclide solubilities.

2.3.2 Calculation of C_0 for Treated Waste

For treated waste, in addition to sorption and solubility constraints, C_0 is controlled by the leach rate of the waste matrix (grout or vitrified waste). Leaching of the waste matrix (and the subsequent release of bound constituents) is a complex process, dependent on the water chemistry at the edge of the waste matrix (which changes as a function of time), diffusion coefficients, matrix and contaminant solubilities, kinetics, and other factors. Several different processes could actually act to leach contaminants from the waste matrix. These include 1) actual dissolution or alteration of the waste matrix, with subsequent release of the bound contaminants; 2) diffusion through the pore water in the waste matrix; and 3) advective migration through the waste matrix. For the purpose of this exercise, it is assumed that infiltrating water migrates along the surface of the fixated or vitrified blocks of waste and advective transport out of the blocks is negligible. Therefore, advective transport was not included in the analytical formulation.

The total rate of release from vitrified and fixated waste was considered equal to the sum of dissolution and diffusion release. The equations below are based on the following assumptions:

1) For dissolution-based leaching, contaminants cannot be released from the waste matrix until the matrix is dissolved or altered in some manner. This dissolution/alteration rate is described by a rate (in mass/area/time) multiplied by a surface area. Contaminants are released congruently (i.e., based on their mass fraction) with the matrix dissolution.

2) Dissolved constituents quickly partition between water and the altered waste matrix according to a partition coefficient, thereby buffering the dissolved concentration.

3) The dissolved concentration is always constrained by the solubility of the contaminant.

This results in the following equations for C_0 for treated waste:

$$C_0 = MIN\left[C_{dl},\ C_{sol},\ \frac{M_w}{(K_{d,aw} + \frac{\theta_{aw}}{\rho_{aw}})}\right] \tag{2-2}$$

where

$$C_{dl} = C_d + C_l \tag{2-3}$$

and

$K_{d,aw}$ = partition coefficient between altered waste matrix and water (L/kg);

ρ_{aw} = bulk density of altered waste matrix (kg/L);

θ_{aw} = moisture content of altered waste matrix (unitless);

C_d = maximum source concentration resulting from matrix dissolution (mg/L); and

C_l = maximum source concentration resulting from diffusion of mass through pore moisture of waste matrix (mg/L).

Equation 2-3 sums the concentrations due to the two release processes (dissolution and diffusion). Expressions for calculating C_d and C_l are derived below. Equation 2-2 ensures that the source concentration is constrained by solubility and partitioning criteria.

The maximum source concentration resulting from matrix dissolution, C_d, is computed by dividing the maximum matrix dissolution rate by the volumetric flow rate through the system, and accounting for partitioning:

$$C_d = \frac{D\,S\,M_w\,(10^{-3})}{I_b w_b L_b \left(1 + \frac{\rho_{aw}}{\theta_{aw}} K_{d,aw}\right)} \qquad (2\text{-}4)$$

where:

D = waste matrix dissolution rate (kg/m^2/yr);

S = waste matrix surface area for one entire trench (m^2);

I_b = infiltration rate through barrier (m/yr);

w_b = trench width (m); and

L_b = trench length (m).

The maximum source concentration resulting from diffusion through pore moisture in the matrix, C_l, is computed by dividing the maximum rate of diffusion (i.e., the diffusive leaching rate) by the volumetric flow rate through the system. At early time (before more than 20 percent of the inventory is depleted), an approximate analytical expression for a diffusive leaching rate is given by ANS (1986) as follows:

$$DR = A_o\, e^{-\lambda t} \left(\frac{S}{V}\right) \sqrt{\frac{D_e}{\pi t}} \qquad (2\text{-}5)$$

where:

DR = rate of diffusive leaching (mg/yr);

A_o = total amount of contaminant mass (mg);

V = waste volume (m^3);

D_e = effective pore water diffusivity (m^2/yr);

t = time (yr); and

λ = decay coefficient (yr^{-1}).

Note that effective diffusivity includes retardation effects. Expressing A_o in terms of the concentration, volume and density of the waste, and conservatively neglecting decay simplifies Equation 2-5 as follows:

$$DR = M_w \, S \, \rho_w \sqrt{\frac{D_e}{\pi t}} \tag{2-6}$$

Equation 2-6 assumes a saturated medium. For unsaturated media, diffusivity decreases as the percent saturation decreases. This is accounted for in the model by substituting $D_e s_w$ for D_e, where s_w equals the percent saturation in the waste matrix. Furthermore, Equation 2-6 shows that the rate of diffusion (DR) varies inversely with time (t), such that at early times the rate is very high, and at very long times the rate is very low. Since this study addressed maximum risk and concentration, a time value of 1 year was substituted into the Equation 2-6. Given the very slow rate of infiltration through the waste, the contact time of the leachate will be on the order of years to decades and the total amount of diffusion release will be an average of the diffusion rate during the contact time. Therefore, using the diffusion rate at 1 year is a conservative assumption. Making these assumptions, and dividing by the volumetric flow rate results in the following equation for the maximum source concentration resulting from diffusion through the matrix:

$$C_l = \frac{M_w \, S \, \rho_w \sqrt{\frac{D_e s_w}{\pi}}}{I_b w_b L_b} \tag{2-7}$$

where:

s_w = percent saturation in the waste matrix (unitless).

Since pore water diffusion is only possible in the presence of moisture and vitrified material has virtually no moisture, this is not a significant release mechanism for vitrified waste.

2.4 CONCENTRATION DIRECTLY BENEATH THE FACILITY (C_1)

C_1 is computed directly as a function of C_0. Assuming plug-flow movement of mass through the liner material (i.e., no longitudinal dispersion), the following equation can be used:

$$C_1 = C_0 e^{-\lambda t_1} \tag{2-8}$$

where:

t_1 = travel time through liner (yrs).

This equation assumes no dilution; therefore, if the contaminant does not decay, $C_1=C_0$.

The travel time through the liner, t_1, is computed by dividing the liner thickness by the transport velocity, and multiplying by the retardation factor:

$$t_1 = \frac{L_l\left(1 + \frac{\rho_l}{\theta_l}K_{d,l}\right)}{MAX\left(\frac{I_b}{\theta_l}, \frac{D_l\, s_l}{L_l}\right)} \tag{2-9}$$

where:

L_l = liner thickness (m);

$K_{d,l}$ = partition coefficient between liner material and water for contaminant (L/kg);

ρ_l = bulk density of liner material (kg/L);

D_l = pore diffusivity of liner material (m^2/yr);

θ_l = moisture content of liner material (unitless);

s_l = percent saturation of liner material (unitless).

The denominator is the maximum of the advective and diffusive velocity. The advective velocity is the rate of migration of a contaminant front assuming plug flow (no diffusion). The diffusive velocity can be thought of as the liner thickness divided by the time required for mass to diffuse through the liner such that the concentration at the far edge is equal to approximately 0.5 of the source concentration. The diffusive velocity to reach 0.1 of the source concentration would be approximately 6 times faster.

2.5 CONCENTRATION AT THE WATER TABLE DIRECTLY BENEATH THE FACILITY (C_2)

C_2 is computed directly as a function of C_1. Assuming plug-flow movement of mass through the unsaturated zone (i.e., no longitudinal dispersion), the following equation can be used:

$$C_2 = DIL_2 C_1 e^{-\lambda t_2} \tag{2-10}$$

where:

DIL_2 = dilution factor for unsaturated zone; and

t_2 = travel time through unsaturated zone (yr).

As illustrated in this equation, the contaminant concentration is affected by both decay and dilution.

The dilution factor and travel time depend on the hydrogeological behavior of the unsaturated zone. In particular, we are interested in the degree to which water infiltrating between the trenches (clean water) mixes with water infiltrating through the trenches (contaminated water). If we assume that there is no mixing, then the dilution factor, DIL_2, is equal to one and the travel time, t_2, is computed as follows:

$$t_2 = \frac{L_u \, \theta_u \left(1 + \frac{\rho_u}{\theta_u} K_{d,u}\right)}{I_b} \tag{2-11}$$

where:

L_u = unsaturated zone thickness (m);

$K_{d,u}$ = partition coefficient between unsaturated zone soils and water (L/kg);

ρ_u = bulk density of unsaturated zone material (kg/L); and

θ_u = average moisture content of the unsaturated zone (unitless).

Assuming some mixing between the contaminated and clean water infiltrating through the unsaturated zone, the dilution factor and travel time are computed as follows:

$$DIL_2 = \frac{I_b \, w_b}{I_b \, w_b + I_s \, w_s \, f_{mix}} \tag{2-12}$$

and

$$t_2 = \left[\frac{\theta_u(d_m - d_t)}{I_b} + \frac{\theta_u(d_u - d_m)}{I_{ave}}\right]\left[1 + \frac{\rho_u}{\theta_u} K_{d,u}\right] \tag{2-13}$$

where:

$$I_{ave} = \frac{I_b w_b + I_s w_s}{w_b + w_s} \tag{2-14}$$

and

I_{ave} = average infiltration rate through unsaturated zone (m/yr);

I_s = average infiltration rate in area of facility between trenches (m/yr);

w_b = trench width (m);

d_t = depth of trench (m);

d_m = mixing depth (m);

d_u = depth to water table (m);

w_s = distance between trenches (m); and

f_{mix} = mixing factor (unitless fraction between 0 and 1).

The mixing factor, f_{mix}, quantifies the degree of mixing in the vadose zone between contaminated leachate and uncontaminated water that infiltrates between the trenches. If $f_{mix} = 0$, there is no mixing; if $f_{mix} = 1$, there is complete mixing. The mixing depth represents the point in the vadose zone where mixing occurs between contaminated infiltration and clean infiltration. Conceptually, this depth corresponds to a lithologic contrast where horizontal migration would likely occur. The travel time calculation assumes that the migration rate is determined by the barrier infiltration rate above the mixing depth and by the weighted average of the barrier and natural infiltration rates below the mixing depth. As a result, the travel time is reduced as the mixing depth moves closer to the bottom of the trench. In contrast, the dilution due to mixing is the same no matter what the mixing depth. In reality, any dilution and increased migration rates due to infiltration outside the foot print of the barrier will likely occur in multiple increments at distinct lithologic changes. The simplified approach utilized in this exercise is sufficient considering that the compliance point is at the water table.

2.6 CONCENTRATION IN GROUNDWATER AT THE ERDF BOUNDARY (C_3)

C_3 is computed directly as a function of C_2. Assuming plug-flow movement of mass through the saturated zone (i.e., no longitudinal dispersion), the following equation can be used:

$$C_3 = DIL_3 C_2 e^{-\lambda t_3} \tag{2-15}$$

where:

DIL_3 = dilution factor for saturated zone; and

t_3 = travel time through saturated zone (yr).

As illustrated in this equation, the contaminant concentration is affected by both decay and dilution.

The dilution factor and travel time depend on the hydrogeological behavior of the saturated zone. In particular, the dilution factor is determined by the extent that contaminated water at the surface of the aquifer is mixed with deeper clean water. This is, to a large extent, dependent on the assumptions made regarding the depth and pumping rate of the well through which individuals are exposed to concentration C_3. In this exercise, we assume DIL_3 is computed as follows:

$$DIL_3 = \frac{L_b \left(I_b\, w_b + I_s\, w_s\right)}{(w_b + w_s)\, K\, i\, d_{mix} + L_b \left(I_b\, w_b + I_s\, w_s\right)} \tag{2-16}$$

where:

K = hydraulic conductivity of saturated zone (m/yr);

i = hydraulic gradient of saturated zone (unitless); and

d_{mix} = mixing depth in saturated zone (m).

The travel time through the saturated zone, t_3, is computed as follows:

$$t_3 = \frac{L_s\, n_s \left(1 + \frac{\rho_s}{n_s} K_{d,s}\right)}{K\, i} \tag{2-17}$$

where:

L_s = distance from edge of facility to nearest trench perpendicular to direction of groundwater flow (m);

$K_{d,s}$ = partition coefficient between saturated zone material and water (L/kg);

ρ_s = bulk density of saturated zone material (kg/L);

n_s = effective porosity of saturated zone (unitless);

K = effective hydraulic conductivity of saturated zone (m/yr); and

i = hydraulic gradient in saturated zone (unitless).

2.7 EXPOSURE ASSESSMENT AND RISK CHARACTERIZATION

Either lifetime incremental cancer risk (ICR) or hazard quotient (HQ, an indicator of non-carcinogenic toxic effects) is calculated based on concentrations of the six contaminants at the three compliance points. Expressing performance in terms of risk allows combining the effects of multiple contaminants into two parameters (ICR and HQ) and also illustrates the general magnitude of potential health effects due to the ERDF. Risk calculations were performed using the approach described in the Hanford Site Baseline Risk Assessment Methodology (DOE-RL 1993).

2.7.1 Exposure Assessment

The exposure assessment assumed residential exposure to groundwater at each point of compliance. Other exposure pathways, such as direct exposure and soil ingestion, were not considered, primarily because these are not realistic pathways given the institutional controls at the ERDF. As discussed in Section 1.3, residential exposure to groundwater at the three points of compliance does not reflect current or proposed land use. Therefore, the risk estimates provided in this study should not be considered actual risk estimates, but simply as relative measures of system performance.

The possible intake mechanisms for residential exposure to groundwater are: ingestion, dermal, and inhalation. These mechanisms are not operative for all the contaminants. For example, inhalation uptake is only applicable for volatile constituents (i.e., TCE) and dermal uptake is only operative for PCBs, TCE, and Cr(VI). Ingestion is applicable for all the constituents. Constituent specific summary intake factors and permeabilities (necessary for the dermal pathway) are provided in Table 2-1. These summary intake factors were obtained from Appendix A of the HSBRAM (DOE-RL 1993). The dermal summary intake factors must be multiplied by constituent-specific permeability factors (provided in Tables 2-2 and 2-3) to obtain constituent-specific summary intake factors.

2.7.2 Toxicity Assessment

With the exception of Cr(VI), all the simulated contaminants are carcinogens. For carcinogens, toxicity is represented by a slope factor and health effects are expressed in terms of an ICR. Slope factors for each of the applicable uptake mechanisms are provided in Table 2-2.

Because Cr(VI) is a chemical toxin but not an oral carcinogen, it was necessary to express Cr(VI) health effects in terms of a HQ. For non-carcinogens, toxicity is represented by a reference dose (RfD). The RfD for Cr(VI) is provided in Table 2-3.

2.7.3 Risk Characterization

Risks are calculated using the intake factors provided in the exposure assessment (Section 2.7.1), the toxicity factors provided in the toxicity assessment (Section 2.7.2), and the calculated groundwater concentrations. The intake factors and toxicity factors are combined into chemical-specific conversion factors that are multiplied by the groundwater concentration to calculate risk. The equation for determining the ICR conversion factors is:

$$CF_{ICR} = IF\ SF\ P \tag{2-18}$$

where:

CF_{ICR} = Conversion Factor for ICR (L/mg or L/pCi);

IF = Summary Intake Factor;

SF = Slope Factor $(mg/kg\text{-}d)^{-1}$;

P = Permeability (only applicable to dermal uptake) (cm/hr).

The equation for determining the HQ conversion factors is:

$$CF_{HQ} = \frac{IF\ P}{RfD} \tag{2-19}$$

where:

CF_{HQ} = Hazard Quotient Conversion Factor (L/mg or L/pCi);

IF = Summary Intake Factor;

RfD = Reference Dose (mg/kg-d);

P = Permeability (only applicable to dermal uptake) (cm/hr).

Constituent-specific conversion factors calculated using Equations 2-18 and 2-19 are provided in Table 2-4. The ICR and HQ are calculated by multiplying the groundwater concentration by the appropriate conversion factors:

$$\text{ICR} = \text{C}\ \text{CF}_{\text{ICR}} \tag{2-20}$$

and

$$\text{HQ} = \text{C}\ \text{CF}_{\text{HQ}} \tag{2-21}$$

where:

C = groundwater concentration (mg/L or pCi/L).

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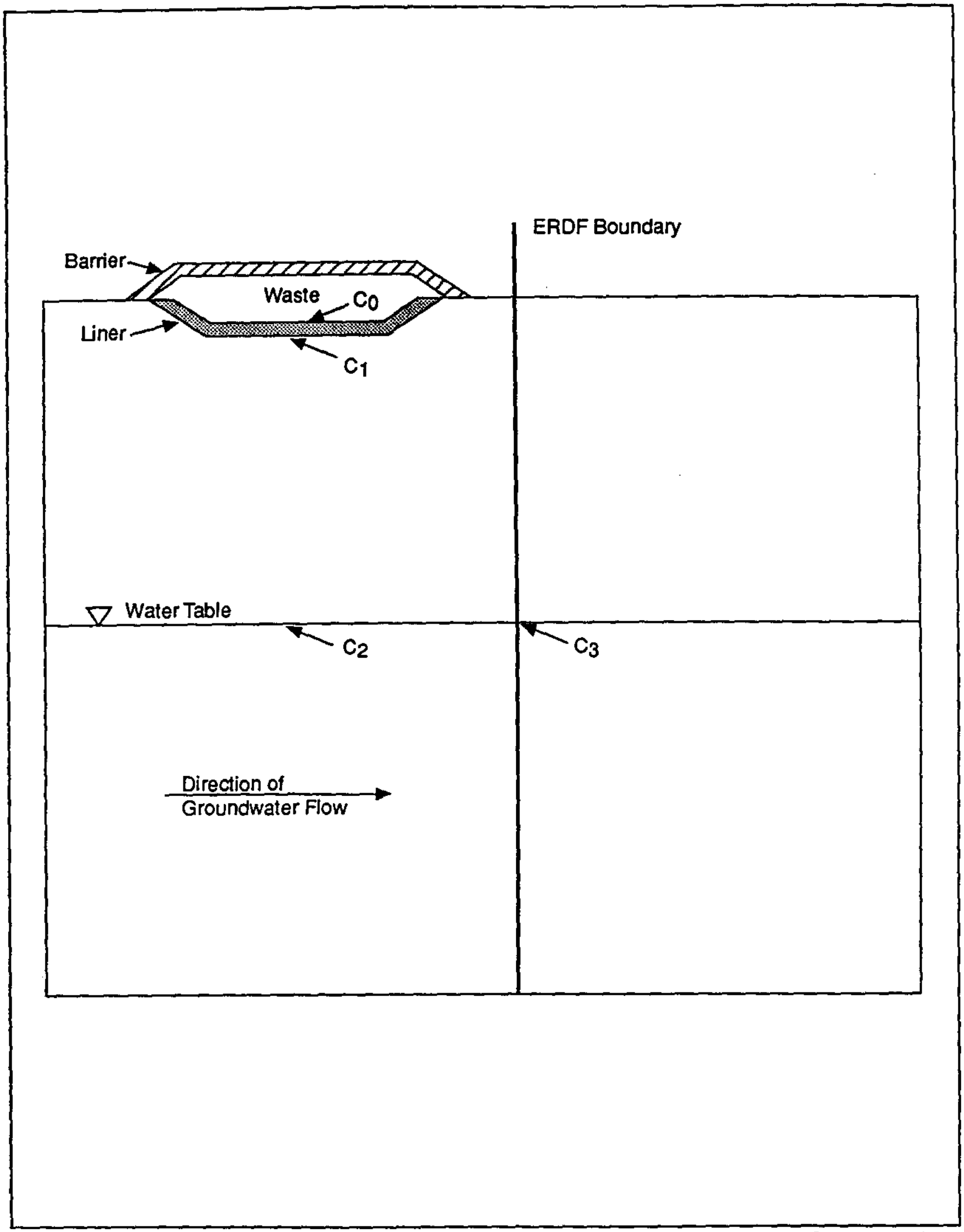

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Figure 2-1. Locations of Compliance Points.

Table 2-1. Summary Intake Factors for Residential Exposure to Groundwater.

Contaminant Class	Ingestion	Inhalation	Dermal
Non-carcinogens	6.2E-02	(not used)	4.9E-02
Non-radioactive carcinogens	1.2E-02	4.6E-02	2.1E-02
Radioactive carcinogens	2.2E+04	(not used)	(not used)
Source: HSBRAM (DOE-RL 1993)			

Table 2-2. Slope Factors and Dermal Permeability for Carcinogens.

Contaminant	Ingestion Slope Factor	Inhalation Slope Factor	Adjusted Dermal Slope Factor	Dermal Permeability
Radionuclides	$(pCi)^{-1}$			
Uranium-238	2.8E-11[a]	(NA)	(NA)	(NA)
Strontium 90	3.6E-11[a]	(NA)	(NA)	(NA)
Plutonium-239	2.3E-10[a]	(NA)	(NA)	(NA)
Non-Radioactive	$(mg/kg\text{-}d)^{-1}$	$(mg/kg\text{-}d)^{-1}$	$(mg/kg\text{-}d)^{-1}$	(cm/hr)
Trichloroethene	1.1E-02[b]	6.0E-03[b]	1.1E-02[b]	2.0E-01[c]
PCB (aroclor-1260)	7.7E+00[d]	(NA)	8.6E+00[d]	7.1E-01[c]

[a] HEAST (EPA 1992b)
[b] Superfund Technical Support Center (EPA 1993a)
[c] (EPA 1992a)
[d] IRIS (EPA 1993b)

(NA) Not applicable because the contaminant is not volatile or is not adsorbed dermally.

Table 2-3. RfD Factors and Dermal Permeabilities for Non-carcinogenic Contaminants.

Contaminant	Ingestion RfD (mg/kg-d)	Adjusted Dermal RfD (mg/kg-d)	Dermal Permeability (cm/hr)
Chromium (VI)	5.0E-03[a]	5.0E-04[b]	2.0E-03[c]

[a] Calculated using an RfD of 5.0E-04 (EPA 1993b) and an oral absorption factor of 0.1 (SRC 1989).
[b] IRIS (EPA 1993b)
[c] (EPA 1992a)

Table 2-4. Factors for Converting Groundwater Concentrations to Lifetime Incremental Cancer Risks (ICR) and Hazard Quotients (HQ).

Contaminant	ICR Conversion Factor	HQ Conversion Factor
Radionuclides	**(L/pCi)**	
Uranium-238	6.2E-07	(NA)
Strontium-90	7.9E-07	(NA)
Plutonium-239	5.1E-06	(NA)
Non Radionuclides	**(L/mg)**	**(L/mg)**
Trichloroethene	4.6E-04	(NA)
PCBs (aroclor-1260)	2.2E-01	(NA)
Chromium (VI)	(NA)	1.2E+01
NA = not applicable.		

3.0 MODEL PARAMETERS

To quantify the fate and transport of contaminants, it is essential to evaluate the hydrogeological conditions of the ERDF site, the physical and chemical properties of the waste forms, liners, and barriers, and the fate and transport properties of each contaminant constituent. Due to the scoping nature of this study and the absence of site-specific and waste-specific experimental data, a literature review was conducted to derive appropriate input parameter values. The parameter estimation relied upon Hanford Site background information when available. Non-Hanford Site information was utilized in some cases. The most likely value for each parameter, (and when appropriate the likely range for each parameter) is provided.

3.1 GENERAL PARAMETERS

A number of parameters (referred to as general parameters) are the same for each of ERDF alternative designs. These include the natural infiltration rate, dimensions of trench, and the physical and hydrogeological properties of both vadose zone and saturated zone soils. These parameters are summarized in Table 3-1.

3.1.1 Natural Infiltration Rate

To estimate the natural infiltration rate at the ERDF site, information from a variety of lysimeter and modeling studies was evaluated. The longest running lysimeter study was conducted using a pair of lysimeters (one open-bottom, the other closed-bottom) installed in the 200 East area in 1971. Moisture content data from these lysimeters indicate a relatively constant moisture content of 6 percent below a depth of 5 m (17 ft). An analysis of the unsaturated hydraulic conductivity of the lysimeter soils (which were primarily sands) suggested an infiltration rate of approximately 0.5 cm/year (0.2 in./year) (Jones 1978). Coring of the closed-bottom lysimeter in 1985 revealed little change in moisture content below a depth of 3 m (10 ft) and no accumulation of moisture in the bottom of the lysimeter. Routson et al. (1988) concluded that the infiltration rate at this location was negligible (less than 0.2 cm/year). Deep-rooted tumbleweeds and other vegetation are believed to have been present on the 200 Area lysimeter for much of the study period. UNSAT-H modeling of the closed-bottom lysimeter indicated that the rate of infiltration was primarily controlled by the surface vegetation; infiltration was much higher when transpiration due to vegetation was eliminated from the model.

Lysimeters have been installed at a variety of other facilities (such as the Buried Waste Test Facility). As summarized in Gee et al. (1992) infiltration rates for these lysimeters range from 0 to 20 cm/yr (8.0 in./yr) and illustrate a strong dependance on soil type and vegetation type. With the exception of one lysimeter which had an infiltration rate of 1.0 cm/yr (0.4 in./yr), no infiltration occurred in lysimeters with deep-rooted vegetation (Gee et al. 1992). Based on these results, a natural infiltration rate between the trenches of 0.5 cm/yr (0.2 in./yr) was used for the ERDF simulations. The uncertainty analysis utilized a range from 0 to 5 cm/yr (2.0 in./yr).

3.1.2 Vadose Zone Parameters

The range of moisture content in 200 Area soils of the Hanford formation is 2 percent to slightly over 6 percent (Last et al. 1989). Data from the 200-East area lysimeters indicate soil moisture values of 3 percent and under, to a depth of 18.3 m (60 ft) (Gee, 1987). The most likely value selected for modeling purposes was 4.5 percent with a range of 2 to 6 percent. These values should be representative of probable soil moisture contents at the ERDF site.

A geologic cross section of the northern edge of the proposed ERDF site is shown in Figure 3-1. The ground elevation across the proposed ERDF site ranges from approximately 200 m (660 ft) to 230 m (760 ft). As shown in Figure 3-2, the water table elevation ranges from approximately 140 m (460 ft) to 120 m (400 ft). The vadose zone thickness ranges from approximately 70 m (230 ft) to 100 m (330 ft), and is about 80 m (260 ft) thick in the center of the ERDF site. The value of 80 m (260 ft) is a good average representation of the vadose zone thickness at this site, and was used for modeling purposes.

The vadose zone mixing depth is also based on the geologic cross section provided in Figure 3-1. The Plio-Pleistocene unit, which has a lower permeability than the rest of the vadose zone materials and may encourage horizontal migration, is found in the western portion of the ERDF site at a depth of approximately 50 m (165 ft). Therefore, a most likely depth of 50 m (165 ft) and a range of 30 m to 80 m were used in the simulations.

The vadose zone mixing factor was assumed to equal 0.25, which corresponds to dilution of the contaminated infiltration through the trenches by 25 percent of the clean infiltration between the trenches. A range of 0 to 1.0 was used for the uncertainty analyses. The dry density of soil in the vadose zone was assumed to equal 1.6 kg/L.

3.1.3 Saturated Zone Parameters

The saturated hydraulic gradient was estimated based on the water table elevation shown on Figure 3-2. The gradients at the ERDF range from 0.0045 along the northern boundary of the site to 0.0025 along the southern boundary. The best estimate gradient selected (0.0035) represents a value of the gradient just over half of a mile south of the northern boundary of the ERDF. A range of 0.001 to 0.005 was used for the uncertainty analyses.

The saturated hydraulic conductivity of the uppermost aquifer unit was estimated based on pump test results from the 200 West Area (Last et al. 1989). The value was selected after a comparison was made of the water table map of the ERDF site and a 200 West Area water table map. It was determined that the hydraulic gradient near the ERDF site was similar to the hydraulic gradient near the test wells in the 200 West Area. Assuming that the fluxes of water through these areas were similar, the hydraulic conductivities reported for the 200 West Area should adequately represent the ERDF area. The best estimate value used in the modeling (30 m/d or 100 ft/d) represents the approximate middle of the range of hydraulic conductivities for the 200 West Area. A range of 1 m/day to 100 m/day was used for the uncertainty analyses.

The best estimate saturated zone porosity was assumed to equal 0.4 with a range of 0.3 to 0.5 used for the uncertainty analyses. The dry density was assumed to equal 1.6 kg/L. The best estimate saturated zone mixing depth was assumed to equal 5 m, with a range of 1 m to 10 m used for the uncertainty analyses.

3.1.4 ERDF and Trench Dimensions

Cross-sections of the proposed trench dimensions are shown in Figure 3-3. The trench width is 90 m (300 ft) at the ground surface and 30 m (100 ft) at the base of the trench. For unlined trenches, the trench sides can be cut at a 1:1.5 (vertical:horizontal) slope which allows a depth of 20 m (66 ft). For lined trenches, the trench slopes must be 1:3 (vertical:horizontal) to allow placement of the clay liner; consequently, the depth of the lined trenches is reduced to 10 m (33 ft) with a corresponding reduction in cross-sectional area.

The concrete vault was assumed to have the same width dimensions as the trenches and a depth of 7.5 m (25 ft). These width dimensions are significantly different than the proposed vault dimensions, but were necessary to simplify the uncertainty analysis. These differences should not change the results by more than a factor of 2.

The trench length was assumed to equal 3,000 m (10,000 ft), which is several hundred meters less than the approximate width of the initial phase of the ERDF facility (see Figure 1-2). The trench separation was assumed to equal 100 m (330 ft) and the smallest distance between the trench and the ERDF boundary was assumed to equal 100 m (330 ft).

3.2 BARRIER PARAMETERS

The modified RCRA barrier consists of seven layers (Figure 1-5), and is designed to prevent water infiltration, biological intrusion, and resist erosion. The top layer consists of McGee Ranch silt with pea gravel admix to help resist erosion. The second layer consists of compacted McGee Ranch silt. Both of these layers are intended to support vegetative growth to enhance evapotranspiration. The third layer consists of a compacted sand (a sand filter), designed to prevent downward migration of the fine soil from the top two layers. The fourth layer is a gravel layer intended to act as a filter to prevent downward migration of the sand from the third layer. Layer five consists of coarse gravel, and is intended to function as a lateral drainage layer. Layer six is comprised of a surface treated asphaltic concrete, which is designed to prevent moisture and biological intrusion into the waste below. At this time, there are no empirical data available regarding the hydrological performance of the modified RCRA barrier, so analyses were conducted by examining lysimeter data and modeling results, and exercising professional judgement.

The Hanford barrier consists of nine layers (Figure 1-4), and is designed to prevent water infiltration, biological intrusion, and resist erosion. The top four layers are made of the same materials (but with greater layer thicknesses in some cases) as the corresponding layers of the RCRA barrier, and are designed for the same purposes. Layer five consists of crushed basalt, and is designed to prevent biological intrusion. Layer six is a coarse gravel designed to act as a lateral drainage layer. Layer seven is comprised of a surface treated asphaltic concrete, designed to prevent moisture and biologic intrusion into the waste below. At this time, there are no empirical data available regarding the hydrologic performance of the Hanford barrier, so analyses were conducted by examining lysimeter data and modeling results, and exercising professional judgement.

As discussed in Section 3-1, lysimeter studies suggest that no infiltration will occur through vegetated, silty, surface soils. This is supported by UNSAT-H computer modelling which suggests that no infiltration will occur through either engineered barrier (Nichols 1991). The UNSAT-H modeling effort analyzed a simple system consisting of a vegetated top layer of unmodified McGee Ranch silt with a thickness of 0.7 m (2.3 ft), and a second layer consisting of 15 cm (6.0 in.) of fine sand. Although this modeling did not use the exact soil parameters and layer dimensions found in the modified RCRA and Hanford barriers, it demonstrates the ability of vegetated McGee Ranch silt to minimize infiltration.

As described in Appendix A, computer modeling was also conducted with the HELP 2.05 model to address the sensitivity of each barrier model to soil parameters, evapotranspiration depths, and extreme precipitation events. This modeling showed that infiltration could be significantly affected by the depth of the rooting zone in relation to the thickness of the first layer. If the rooting zone did not penetrate the entire thickness of the first layer, the infiltration rate was as high as 0.15 cm/yr (0.06 m/yr). When the rooting zone did penetrate the first layer the infiltration rate was less than 1×10^{-3} cm/yr (4×10^{-4} in./yr). In reality, the division between the top two layers (which consist of virtually the same material with different compactions) will be less well defined than assumed in the modeling; this is because some compaction of the top layer will occur due to equipment traffic during construction. Assuming the evaporation zone extends through the upper uncompacted silt layer, the modeling indicated very low infiltration (less than 1×10^{-3} cm/yr) through both the modified RCRA barrier and the Hanford barrier for existing conditions at Hanford. This was true even when hypothetical extreme precipitation events were added to the simulated climate conditions. Assuming that performance of the barriers will degrade somewhat with time and allowing for minor climatic changes that may occur in the future, the best estimate infiltration rate was assumed to equal 0.03 cm/year (0.01 in./yr) for the Hanford Barrier and 0.06 cm/yr (0.02 in./yr) for the modified RCRA barrier. A range of 0.001 cm/yr to 0.1 cm/yr was assumed for the uncertainty analyses. For the uncertainty analysis, the infiltration rates for both the Hanford barrier and the modified RCRA barrier were assumed to be correlated to natural infiltration with a 0.85 correlation coefficient.

It was assumed for the "no barrier" alternative that the waste is buried under several feet of soil with characteristics similar to the soils found in the ERDF area. The infiltration values used for the "no barrier" alternative were assumed to be the same as the natural infiltration rate between the trenches. For the uncertainty analysis, the infiltration rate for the "no barrier" alternatives was assumed to be correlated to natural infiltration between the trenches with a 0.95 correlation coefficient.

3.3 LINER PARAMETERS

Four types of liners were included in the alternatives: no liner, a single liner, a MTR liner, and a concrete vault. The MTR liner is illustrated in Figure 1-3. The MTR liner consists of a 60 cm (24 in.) thick soil protection layer over two flexible membrane liners (along with associated drainage layers) on top of a 0.9 m (3.0 ft) thick compacted clay layer. The single layer is the same as the MTR liner except it only has one flexible membrane liner. The protection layers were not included in the modelled liners, based on the assumption that leachate from the waste would migrate through the protection layers during the operational period. In addition, the flexible membrane liners were also not included in the modelled liners based on the assumption that these synthetic materials would degrade relatively quickly. Although this assumption is probably reasonable, very little information is available concerning the durability of synthetic liners in arid environments. Since the synthetic liners were not included, the single barrier performed exactly the same as the MTR liner. The concrete vault was modeled as a grout slab with a thickness of 0.5 m (1.7 ft). Similar to the flexible membrane liners, the asphalt coating on the vault was assumed to degrade and was not simulated.

Liner parameters are provided in Table 3-2. As discussed in Section 3.1.4, the trench heights are different for different liner types due to the allowable side-wall slopes. The liner thickness only accounts for the silt/bentonite layer, or the concrete thickness in the case of the vault. Assuming a thickness of zero for the no-liner case eliminates any liner effects and consequently C_1 equals C_0. The bulk density is assumed to equal 1.5 kg/L for the silt/bentonite layer and 2.4 kg/L for the concrete in the vault. Porosity and percent saturation are typical for liners in arid environments.

Soil/water partitioning (K_d) factors are generally higher in fine-grained materials than in coarse-grained materials. Therefore, the liner travel time equations utilize the vadose zone K_d multiplied by a liner-specific adjustment factor. This factor was assumed to equal 5 for the silt/bentonite liners and 1 for the vault.

Because different constituents diffuse at different rates, liner diffusivities are discussed under constituent-specific parameters (Section 3.5). Diffusivities through the concrete vault were assumed to be three orders of magnitude less than in the silt/bentonite layers.

3.4 WASTE RELEASE PARAMETERS

As discussed in Section 2.3, waste release is controlled by solubility and sorption limitations, matrix dissolution, and diffusion. Because they are constituent-specific, solubilities and diffusion coefficients are discussed in Section 3.5. Sorption is assumed to be controlled by the same parameters (K_d, moisture content, and soil density) as in the vadose zone. Although this is probably reasonable for the untreated soil waste, it is less reasonable for the fixated and vitrified waste which will be altered by the treatment process. Given the lack of information regarding sorption in these treated waste forms, use of vadose zone parameters was a necessary simplification.

3.4.1 Physical Characteristics of Waste Type

Matrix block dimensions, density, and moisture content for each waste type are summarized in Table 3-3. The matrix block dimensions are important because they determine the surface area of matrix exposed to infiltrating water. As the size of the matrix blocks increases, the exposed surface area decreases and the rates of matrix dissolution and constituent diffusion decrease. The surface area was calculated assuming the blocks of waste were cubes. The assumed side-length for the cubes was 1.0 m (3.3 ft) for fixated waste and 0.5 m (1.7 ft) for vitrified waste. These dimensions were assumed to represent either the size of the matrix blocks when placed in the trench or the average matrix block size due to fracturing.

3.4.2 Matrix Dissolution Rate

The leach rate of vitrified material when exposed to deionized water at 90°C has been reported to equal 5E-02 kg/m^2/yr (Nelson et al. 1991). In another experiment (performed at a temperature of 25°C and a pH of 8.0), a dissolution rate of 4E-05 kg/m^2/yr was measured (Knauss et al. 1990). These differences are likely due in part to differences in temperature. A best estimate dissolution rate of 3.6E-03 kg/m^2/yr with a range of 3.6E-05 kg/m^2/yr to 1.0E-02 kg/m^2/yr was selected for vitrified material.

Published dissolution rates for fixated wastes were not located. This is because dissolution rates of fixated wastes are affected by a variety of factors such as chemical composition of waste, the pH and redox potential of the infiltrating water, chemical reactions, adsorption, pressure, and temperature. A best estimate dissolution rate of 0.1 kg/m^2/yr, which is two orders of magnitude greater than that of vitrified waste, was assumed for fixated waste. A range of 0.01 kg/m^2/yr to 0.5 kg/m^2/yr was assumed for the sensitivity analyses. Dissolution rates are summarized in Table 3-3.

3.5 CONSTITUENT-SPECIFIC PARAMETERS

Constituent-specific parameters include waste diffusivity, liner diffusivity, soil/water partitioning coefficient (K_d), decay or degradation rate, and solubility. The values of these parameters used in the modeling are summarized in Table 3-4 and are discussed below.

3.5.1 Waste Diffusivity

A porosity of zero was assumed for the vitrified waste form; therefore, diffusion is not a release mechanism for vitrified waste.

Diffusion is the most important mass transport mechanism for fixated waste (Alcorn et al. 1992). The effective diffusion coefficients for reactive constituents account for retardation as well as the physical hindrance to leaching caused by the small pore sizes and tortuosity of the cement matrix (Serne and Wood 1990). Serne and Wood (1990) suggest that the following values of effective diffusivity should be used for generic grout performance assessment calculations when actual grout-specific data are lacking:

- Chromium - 3.2E-07 (m^2/yr)
- Strontium - 1.6E-07 (m^2/yr)
- Uranium - 3.2E-09 (m^2/yr)
- Plutonium - 1.6E-09 (m^2/yr)

Effective diffusivities of 3.2E-07 (m^2/yr) for TCE and 1.6E-09 (m^2/yr) for PCBs were also assumed. Values for these organic constituents were selected based on constituents with similar retardation factors (chromium for TCE and plutonium for PCB). Ranges for waste diffusivity used in the uncertainty analysis are provided in Table 3-4.

3.5.2 Liner Diffusivity

Silt amended with bentonite is the proposed to liner material at the ERDF. No fate and transport data are available for this material. However, Brandberg and Skagius (1991) studied the important transport parameters of bentonite, including diffusivities for different radionuclides. The following constituent-specific pore diffusivities (which do not include retardation) were recommended for bentonite:

- Strontium - 3.2 m^2/yr
- Uranium and plutonium - 1.3E-02 m^2/yr
- Most constituents in general - 1.3E-02 m^2/yr

Best estimates and assumed ranges for liner diffusivity are provided in Table 3-4.

Magnuson and Yu (1992) experimentally obtained a general diffusivity of 3.2E-05 m^2/yr for a concrete vault. This value is approximately three orders of magnitude lower than that of bentonite proposed by Brandberg and Skagius (1991). Therefore, constituent-specific pore diffusivities for the concrete vault were assumed to be three orders of magnitude lower than that used for the silt/bentonite liners.

3.5.3 Partitioning Coefficient (K_d)

The partitioning coefficient (K_d) is defined as the ratio of adsorbed chemical concentration in the soil matrix to the aqueous solute concentration. The sorbing solute travels at a linear velocity that is lower than the groundwater flow velocity. Best estimates and assumed ranges for K_d are provided in Table 3-4.

Howard (1990) estimated an octanol/water partitioning coefficient (K_{oc}) of 100 for TCE and Montgomery and Welkem (1990) estimated a K_{oc} range of 275 L/kg to 2.6E+06 L/kg for PCBs. Based on the assumption that:

$$K_d = K_{oc} f_{oc} \tag{3-1}$$

where:

f_{oc} = Organic content;

and assuming that f_{oc} in soil is 0.5 percent, the best estimates of K_d for TCE and PCB are assumed to equal 0.5 L/kg and 25 L/kg, respectively. The ranges were assumed to be 0.1 L/kg to 1.0 L/kg for TCE and 20 L/kg to 100 L/kg for PCBs.

Cr(VI) is assumed to be very mobile and is not adsorbed under Hanford ambient geological environment (Serne and Wood 1990). The assumed best estimate for Cr(VI) K_d was 0.001 L/kg and the range was from 0.0001 L/kg to 0.1 L/kg. However, Adriano (1986) states that soluble Cr(VI) can be converted to insoluble Cr(III) under natural conditions; therefore, the fate and transport of Cr(VI) is highly uncertain.

Sr-90 has been extensively studied at the Hanford Site. Serne and Wood (1990) described investigation of strontium sorption under different conditions. Sr-90 K_d's in

Hanford Site soils generally fall between 0 and 100 L/kg, except in the presence of phosphate or oxalate, in which case K_d is on the order of 200 to 500 L/kg. It is recommended that a value of 5 to 10 L/kg is appropriate, recognizing that in some cases sorption values may be much higher (Serne and Wood 1990). Rhodes (1956) and Routson et al. (1981) suggested a K_d of 7 L/kg for strontium in high salt or organic solutions with neutral pH, and 10 L/kg in low salt and organic solutions with neutral to basic pH. Sewart et al. (1987) and Serne et al. (1989) suggest a K_d of 31 L/kg for strontium. A best estimate of 8 L/kg and a range of 3 L/kg to 31 L/kg were used in the simulations.

No laboratory studies have been published describing uranium sorption in Hanford Site soils, but unpublished work (Wayne Martin at PNL) and U1/U2 pond experience (Delegard et al. 1986) suggest that uranium is not significantly sorbed under ambient Hanford groundwater conditions and slightly acid conditions. Ames and Serne (1991) cited a K_d range of 2 to 2000 L/kg for uranium. A best estimate of 1.0 L/kg and a range of 0.01 L/kg to 5 L/kg were used in the simulations.

Recommended K_d values for plutonium under ambient Hanford soil conditions are 100 to 1000 L/kg (Serne and Wood, 1990) and 21 L/kg (Delegard and Barney 1983). A best estimate of 50 L/kg and a range of 21 L/kg to 100 L/kg were used in the simulations.

3.5.4 Decay (or Degradation) Rate

Best estimates for decay coefficients used in the simulations are provided in Table 3-4. The half-life for an unstable nuclide is the time required for one-half of a given number of atoms to decay. Half-lives and decay coefficients for the radionuclides are readily available. The degradation half-life for organic constituents is the time needed for half of the concentration to be degraded or volatilized (Dragun 1988). The half-life ($T_{1/2}$) and its decay or degradation constant (λ) are related by the following equation (Faure 1977):

$$T_{1/2} = 0.693 / \lambda \tag{3-2}$$

Organic chemicals can be degraded biologically or chemically. Furthermore, in the case of volatile compounds (such as TCE), water concentrations will be reduced due to volatilization. Howard et al. (1991) provided a half-life for TCE of 1 year in soil and 4.5 years in groundwater. Because these half-lives probably reflect ideal conditions, a best estimate half-life of 14 years was utilized.

Dragun (1988) suggests that the biodegradation or disappearance rate of PCBs ranges from 0% to 100% in 7 days. A best estimate half-life of 49 years was chosen for PCBs.

Ranges of half-life for TCE and PCBs were not selected in the simulations because the results were not sensitive to these parameters.

Although Cr(VI) is not degradable and does not radioactively decay, it can be reduced to Cr(III). No information was obtained regarding the reduction of Cr(VI) to Cr(III) under natural conditions at Hanford. Therefore, a decay rate of zero was assumed.

3.5.5 Solubility

Solubilities for most metals are a function of the controlling solids and are highly dependent upon physico-chemical parameters such as pH, Eh, and the concentrations of other ionic constituents. The following information was located regarding solubility of the simulated contaminants:

Cr(VI):

- Greater than 1 mg/L at neutral to basic pH (Serne and Wood 1990)
- 0.2 mg/L (Conner 1990).

Sr-90:

- Greater than 1 mg/L at neutral to basic pH (Serne and Wood 1990)
- Less than 25 mg/L (Ames and Serne 1991)

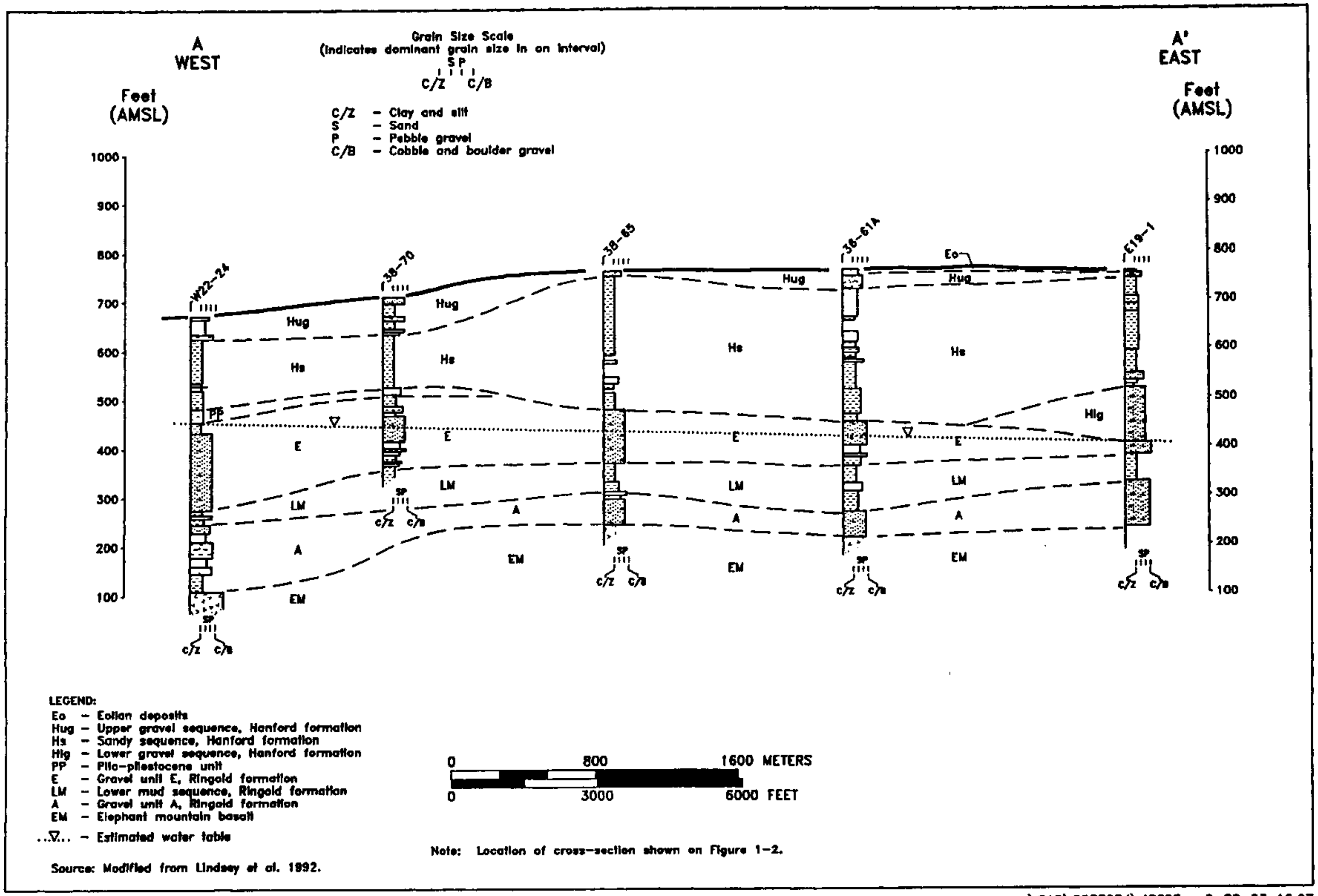
Uranium:

- Greater than 1 mg/L at neutral to basic pH (Serne and Wood 1990)
- Less than 25 mg/L (Ames and Serne 1991)

Plutonium

- Less than 1 mg/L (Serne and Wood 1990)
- Less than 1 mg/L (Ames and Serne 1991)

In comparison with inorganic metals, solubilities for organic compounds are relatively insensitive to changes in water chemistry (except when multiple organic compounds are involved and they begin to behave as co-solvents). Montgomery and Welkem (1990) and Howard (1990) state that the solubility of TCE in water at 25°C is 1100 mg/L. Solubility for the different PCBs range from 0.0027 mg/L to 1.45 mg/L with an average of 0.4 mg/L (Montgomery and Welkem 1990). Based on this information, the best estimates and likely ranges used in the uncertainty analyses are provided in Table 3-4.



Source: Modified from Lindsey et al. 1992.

Note: Location of cross-section shown on Figure 1-2.

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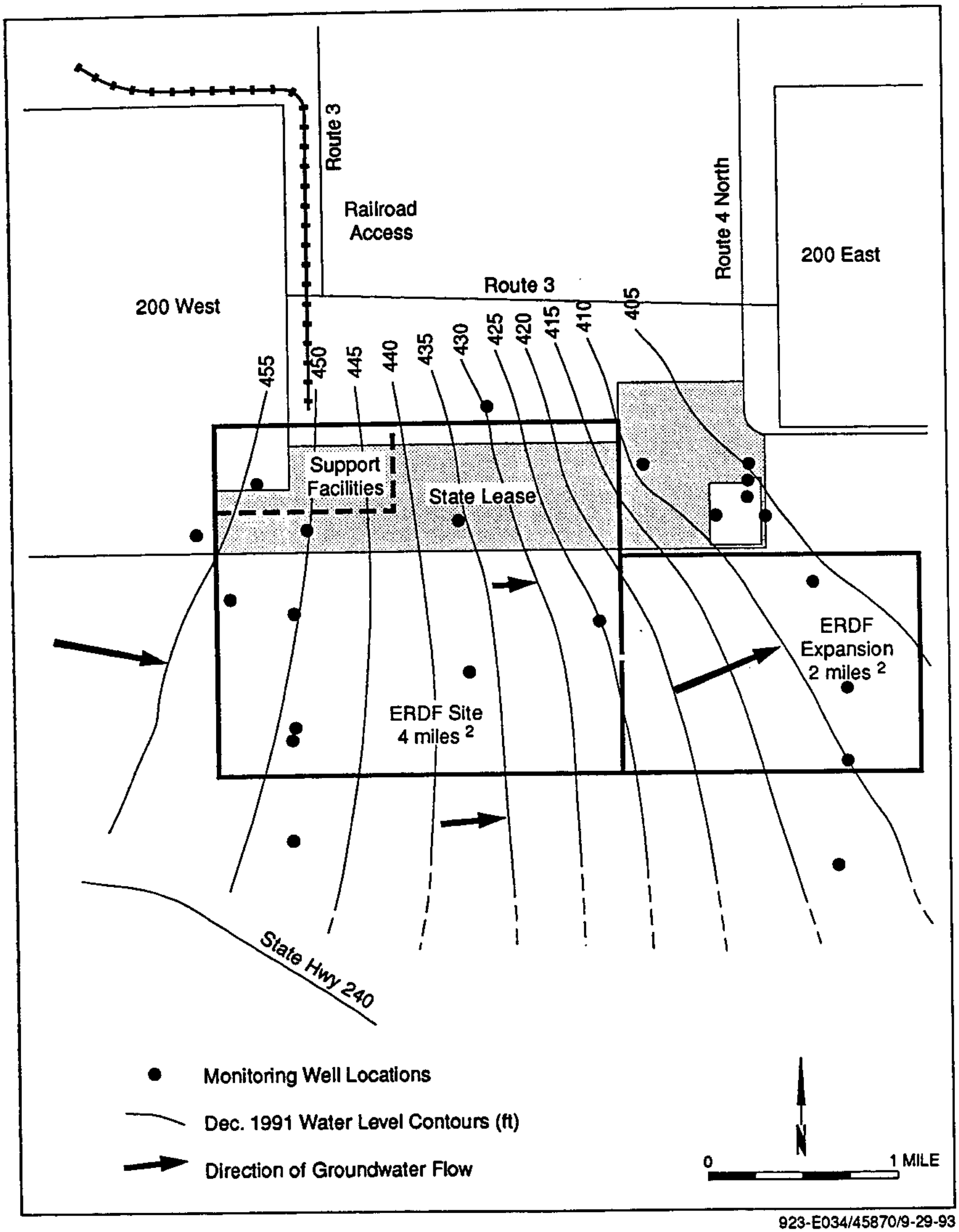
Figure 3-1. Geologic Cross Section at A-A'.



923-E034/45870/9-29-93

Figure 3-2. Water Table Elevations.

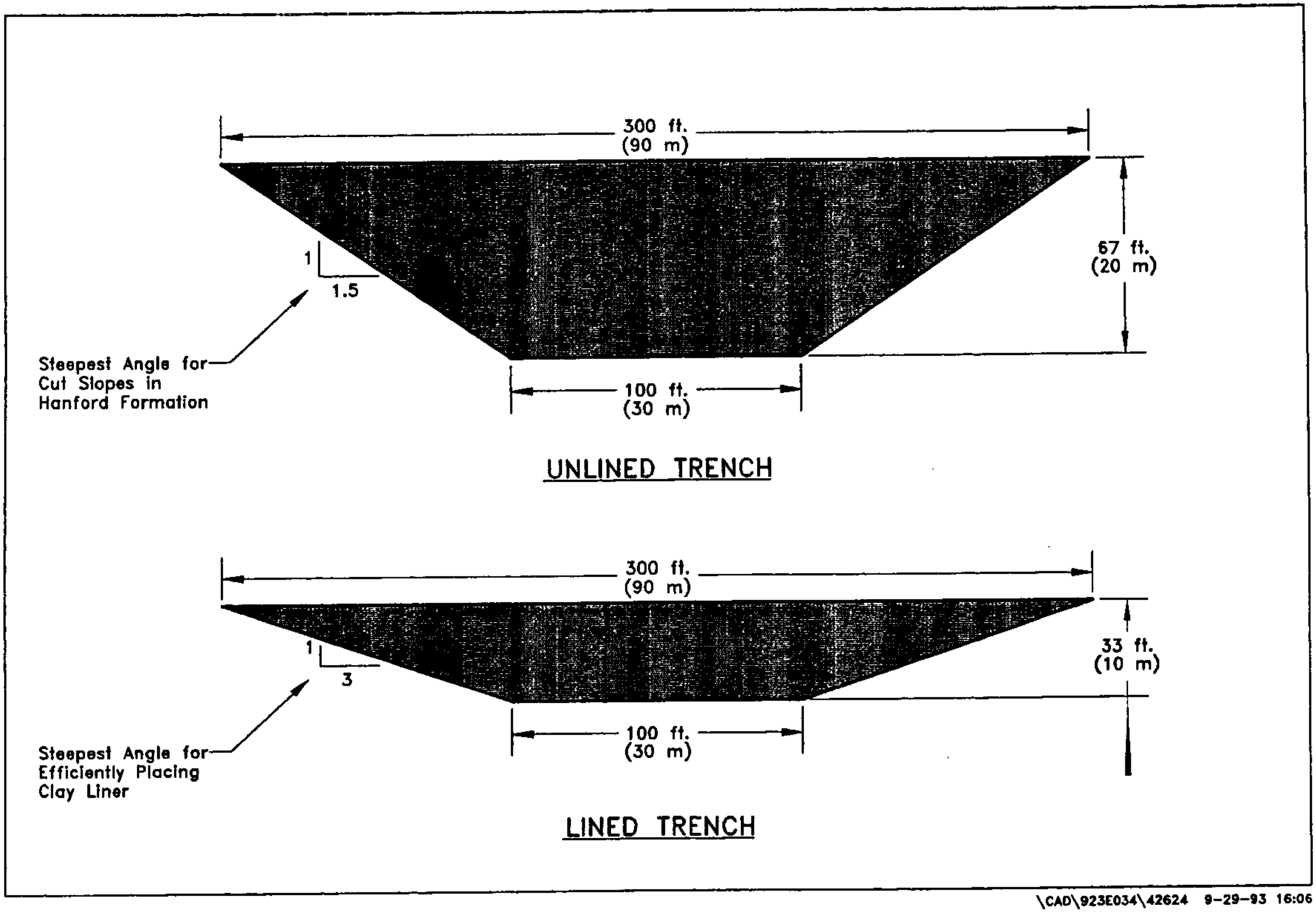


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Figure 3-3. Cross-Sectional Dimensions for Lined and Unlined Trenches.

Table 3-1. General Parameters and Barrier Parameters Used for the ERDF Modeling.

Parameter	Most Likely Value	Range
Upper Trench Width	90 m	Assumed Constant
Lower Trench Width	30 m	Assumed Constant
Trench Length	3000 m	Assumed Constant
Trench Separation	100 m	Assumed Constant
Distance to ERDF Boundary	100 m	Assumed Constant
Vadose Zone Water Content	0.045	0.02 to 0.06
Vadose Zone Thickness	80 m	Assumed Constant
Vadose Zone Mixing Depth	50 m	30 to 80 m
Vadose Zone Mixing Factor	0.25	0 to 1.00
Saturated Zone Porosity	0.40	0.30 to 0.50
Saturated Zone Hydraulic Conductivity	30 m/d	1 to 100 m/d
Saturated Zone Hydraulic Gradient	0.0035	0.001 to 0.005
Saturated Zone Mixing Depth	5 m	1 to 10 m
Soil Density (Dry)	1.6 kg/L	Assumed Constant
Natural Infiltration Rate (also used for no-barrier scenarios)	0.5 cm/yr	0 to 5.0 cm/yr
Hanford Barrier Infiltration Rate	0.03 cm/yr	0.001 to 0.1 cm/yr
Modified RCRA Barrier Infiltration Rate	0.06 cm/yr	0.001 to 0.1 cm/yr

Table 3-2. Liner Parameters.

Parameter	No Liner	Single Liner	MTR Liner	Vault
Trench Height (m)	20	10	10	7.5
Liner Thickness (m)	0	1.0	1.0	0.5
Bulk Density (kg/L)	1.5	1.5	1.5	2.4
K_d Adjustment Factor	1	5	5	1
Porosity	0.50	0.45	0.45	0.05
Percent Saturation	0.50	0.50	0.50	0.10

MTR - Minimum Technology Requirements.

Table 3-3. Waste Form Parameters.

Parameter	Untreated Waste	Fixated Waste	Vitrified Waste
Dissolution Rate (kg/m^2/yr)	(NA)	0.1 (0.01 to 0.5)	3.6E-03 (3.6E-05 to 0.01)
Matrix Block Dimension (m)	(NA)	1.0	0.5
Bulk Density (kg/L)	1.60	2.40	2.65
Moisture Content	0.05	0.01	0
Note: Parameter ranges are shown in parenthesis.			

Table 3-4. Contaminant-Specific Parameters.

Constituent	Liner Pore Diffusivity (m^2/yr)	Grout Diffusivity (m^2/yr)	Partitioning Coefficient (L/kg)	Half-life (year)	Decay Rate (1/year)	Solubility (mg/L)
TCE	1.3E-02 (1.3E-03 to 1.3E-01)	3.2E-07 (3.2E-08 to 3.2E-06)	0.5 (0.1 to 1.0)	14	0.05	1100
PCBs	1.3E-02 (1.3E-03 to 1.3E-01)	1.6E-09 (1.6E-10 to 1.6E-08)	25 (20 to 100)	49	0.01	0.4 (0.027 to 1.45)
Cr (VI)	1.3E-02 (1.3E-03 to 1.3E-01)	3.2E-07 (3.2E-08 to 3.2E-06)	0.001 (0.0001 to 0.1)		0	5 (0.5 to 10)
Sr-90	3.2 (0.32 to 5)	1.6E-07 (1.6E-08 to 1.6E-06)	8 (3 to 31)	29	2.4E-02	5 (1 to 25)
U-238	1.3E-02 (1.3E-03 to 1.3E-01)	3.2E-09 (3.2E-10 to 3.2E-08)	1 (0.01 to 5)	4.5E+09	1.5E-10	5 (1 to 25)
Pu-239	1.3E-02 (1.3E-03 to 1.3E-01)	1.6E-09 (1.6E-10 to 1.6E-08)	50 (21 to 100)	2.4E+04	2.9E-05	0.5 (0.1 to 1)

Note: Parameter ranges are shown in parenthesis.

4.0 RESULTS

Results are presented both as deterministic best estimates of risk, and as likely ranges to reflect the uncertainty in the results. Some of the ICRs reported below are greater than 1, which is not theoretically possible (cancer risk cannot exceed a probability of 1) but can occur mathematically when the concentrations are beyond the calibration range for the slope factors. The slope factors used in the risk assessment equations are generally not considered valid when the ICR exceeds 1E-02. Therefore, in addition to the reasons discussed in Section 1.3 and 2.7.1, any ICR reported above 1E-02 should not be considered a real risk but simply a relative performance measure.

4.1 DETERMINISTIC RESULTS

The deterministic results relied upon the best estimates of input parameters provided in Tables 3-1 through 3-4. Initial leachate concentrations for each constituent are presented in Tables 4-1, 4-2, and 4-3 for waste E, waste C, and waste A alternatives, respectively. Travel times and risks for each constituent at the three compliance points are provided in Tables 4-4 through 4-10.

4.1.1 Results for E Waste Alternatives

Results for the six waste E scenarios at compliance point C_1 (beneath the trench) are provided in Table 4-4. The maximum ICR ranges from 0.33 for the vitrification scenario (Alternative 6) to 99 for Alternatives 1 and 2. Sr-90 is the primary risk driver in all cases. For the unlined alternatives the travel time for all the constituents is zero years. When the alternative includes a liner (Alternatives 3, 4, and 5) the travel time for Sr-90 is 167 years and is diffusion-controlled. The HQ for Cr(VI) equals 61 for all alternatives. The Cr(VI) travel time through the liner is 159 years and is diffusion-controlled.

Results at compliance point C_2 (at the water table) are provided in Table 4-5. The maximum ICR within 10,000 years is zero for all the alternatives. The maximum ICR for all time ranges from 1.8E-04 for Alternative 2 (Hanford barrier) to 8.1E-04 for the no barrier alternatives (Alternatives 1 and 6). Uranium is the only carcinogen that reaches the water table with an ICR greater than 1E-13. The travel time for uranium to the water table is 20,000 years for Alternatives 1 and 6 (no barriers), 180,000 years for Alternative 2 (Hanford Barrier), and 130,000 years for Alternative 3, 4, and 5 (modified RCRA barrier). The HQ for Cr(VI) ranges from 11 for Alternative 2 to 48 for Alternatives 1 and 6.

Results at compliance point C_3 (at the ERDF boundary) are provided in Table 4-6. The results are similar to those at C_2 with approximately an order of magnitude dilution and additional travel time. The maximum ICR within 10,000 years is zero for all the alternatives. The maximum ICR for all time ranges from 7.7E-06 for Alternative 2 to 5.9E-05 for Alternatives 1 and 6. The HQ ranges from 0.45 for Alternative 2 to 3.5 for Alternatives 1 and 6.

Alternatives 3, 4, and 5 have exactly the same results. Alternatives 3 and 4 are designed the same except Alternative 3 has a MTR liner and Alternative 4 has a single liner.

Since these liners are addressed in the same manner in the model, it is expected that the results would be identical. Alternative 5 is designed the same as Alternative 3 except the waste is fixated. Fixation does not change the results because for these six constituents the initial leachate concentration is controlled either by soil/water partitioning or solubility. The rate of diffusion release from the fixated grout is high enough and the rate of infiltration through the facility is low enough that solubility- or partitioning-limited concentrations are achieved. The advantages of grout are better suited to situations with high rates of water flux around the fixated material (such as would be encountered in the saturated zone).

The results for the Alternative 6 (vitrification and no barrier or liner) indicate that vitrification reduces risk at C_1, where Sr-90 is the risk driver. However, at C_2 and C_3, where uranium is the risk-driver, the results for Alternative 6 are virtually identical to those for Alternative 1 (both of which are no-barrier alternatives).

4.1.2 Results for C Waste Alternatives

Results for the eight waste C scenarios at compliance point C_1 (beneath the trench) are provided in Table 4-7. The maximum ICR ranges from 1.2E-03 for Alternatives 9 and 11 to 1.0E-01 for Alternative 8. The primary risk driver for Alternative 10 (vitrification) and the lined with no vitrification alternatives (Alternatives 9, 11, and 12) is U-238. For the unlined Alternatives (Alternatives 7, 8, 15, and 16) the primary risk drivers are Sr-90 and TCE. Alternative 8 has the highest risk because the waste is soil-washed (this concentrates the waste and results in higher leachate concentrations) and the trench is unlined. For the unlined alternatives, the travel time for all the constituents is zero years. When the alternative includes a liner the travel times for Sr-90 (167 years), TCE (2,700 years), and uranium (5,300 years) are diffusion-controlled. The HQ for Cr(VI) equals 61 for all the alternatives. The Cr(VI) travel time through the liner is 159 years and is diffusion-controlled.

Results at compliance point C_2 (at the water table) are provided in Table 4-8. The maximum ICR within 10,000 years is zero for all the alternatives. For equivalent facilities and treatment, the waste C maximum ICRs for all times are the same as for waste E, with the exception of the vitrification alternative (Alternative 10). This is because without vitrification, the leachate concentrations of uranium (the only risk-driver at the water table) are solubility-limited, and do not depend on the soil concentration for the waste. Solubility is not limiting when the waste is vitrified, and the leachate concentration is dependent upon the rate of glass dissolution and soil concentration. The travel time for uranium is the same for waste C alternatives as for waste E alternatives, and is determined by the barrier infiltration rate. The HQ for waste C alternatives are the same as for comparable waste E alternatives since Cr(VI) leachate concentration is limited by solubility.

Results at compliance point C_3 (at the ERDF boundary) are provided in Table 4-9. The results are similar to those at C_2 with approximately an order of magnitude dilution and additional travel time. The maximum ICR within 10,000 years is zero for all the alternatives. The maximum ICR for all time ranges from 7.7E-06 for the Hanford barrier alternatives (Alternatives 8, 11, 12 and 15) to 5.9E-05 for Alternatives 7 and 16 (which are both no-barrier alternatives). The HQ ranges from 0.45 for the Hanford barrier alternatives to 3.5 for Alternatives 7, 10, and 16.

Similar to the waste E alternatives, fixation of waste C does not reduce the risk compared with no-treatment alternatives. Vitrification reduces the risk at C_1 by an order of magnitude and only slightly (less than 10 percent) at C_2 and C_3.

4.1.3 Results for A Waste Alternatives

Results for the two waste A alternatives at compliance point C_1 (beneath the trench) are provided in Table 4-10. Alternative 13 (the "no action" alternative) is the same as Alternative 7, except no hazardous constituents are included. As a result, the TCE ICR equals zero and the total ICR at C_1 is reduced from 2.1E-02 to 1.2E-02. The primary risk driver for Alternative 13 is Sr-90. Alternative 14 includes vitrification, a modified RCRA barrier, and a vault liner. In comparison to Alternative 10 (the waste C vitrification alternative with no barrier) the maximum ICR decreases from 1.3E-03 to 1.0E-03. The travel time through the vault for the controlling contaminant (U-238) is 2,000 years.

Results at compliance point C_2 (at the water table) are provided in Table 4-10. The maximum ICR within 10,000 years is zero for both alternatives. Results for Alternative 13 are exactly the same as for Alternative 7 (the comparable waste C alternative). This is because the hazardous constituents in the C waste do not affect the total ICR. The ICR for the vitrification alternative (Alternative 14) is approximately one-half of the ICR for Alternative 10 (vitrification with no barrier); in addition, the travel time is increased from 20,000 years to 135,000 years due to the addition of the RCRA barrier.

Results at compliance point C_3 (at the ERDF boundary) are provided in Table 4-10. The results are similar to those at C_2 with approximately an order of magnitude dilution and additional travel time. The maximum ICR within 10,000 years is zero for all the alternatives. Results for Alternative 13 are exactly the same as for Alternative 7 (the comparable waste C alternative), for the same reasons as discussed previously. The ICR for the vitrification alternative (Alternative 14) is reduced by a factor of 5 compared with the ICR for Alternative 10.

4.2 UNCERTAINTY ANALYSES

Preliminary performance assessment models for the sixteen alternatives have been developed to provide initial estimates of the fate and transport of the four radionuclides and two organics from the proposed ERDF Site to the three compliance points. The performance assessment analyses required the assignment of specific values to the operational parameters of the various conceptual and mathematical models used. These parameter values were based upon design information or assigned through expert judgment. The estimation of model parameters is subject to greater errors when few measured data are available.

An uncertainty analysis estimates the uncertainty in a system's performance resulting from the uncertainty in one or more factors associated with the system (NRC 1983). These factors include uncertainties in the conceptual model, the mathematical model, and the input parameters. These analyses can be used to assign priorities to the activities in the site characterization program. The uncertainty analyses conducted for this study focuses on input parameter uncertainty. The following parameters were defined as uncertainty variables:

- Natural infiltration rate between trenches
- Barrier infiltration rates (correlated with natural infiltration rate)
- Vadose zone mixing depth
- Vadose zone mixing factor
- Saturated zone mixing depth
- Saturated zone hydraulic gradient
- Saturated zone hydraulic conductivity
- Soil/water partitioning coefficients
- Solubility of waste consituents
- Dissolution rate for fixated and vitrified waste
- Constituent-specific effective diffusivity in grout
- Constituent-specific pore diffusivity in the liner.

Uncertainty distributions for the input variables are illustrated in Appendix B. Triangular distributions were assigned for all the variable parameters, with the upper and lower bounds equal to the ranges provided in Tables 3-1 through 3-4. The apex of the triangular distributions were defined by the best estimates in Tables 3-1 through 3-4.

The Monte Carlo simulations involved approximately 1,500 realizations to obtain relatively stable result distributions. An example of an output distributions is provided in Figure 4-1. This figure shows the distribution of uranium travel times to the ERDF boundary for Alternative 8. The table of percentiles shown on the figure indicates the travel time for each percentile. As illustrated in the figure, the median travel time (50th percentile) was 236,000 years. This means that 50 percent of the travel times were less than 236,000 years and 50 percent were greater than 236,000 years. The 90 percent certainty range was 53,000 years to 937,000 years. This means that 5 percent of the travel times were less than 53,000 years, five percent were greater than 937,000 years, and the remaining 90 percent were between these values. Out of 1,500 realizations, the minimum travel time was 6,774 years and the maximum travel time was 13 million years.

The best estimate results presented in Section 4.1 do not necessarily correspond to the median value in the Monte Carlo simulations. For example, the median uranium travel time for Alternative 8 (see Figure 4-1) is 236,000 years, while the deterministic best estimate is 182,000 years (see Table 4-9). This difference is due to a variety of factors, including: nature of the input parameter distributions (see Appendix B), the non-linear nature of the equations, and the potential for different controlling factors given different input parameters.

4.2.1 Maximum ICR and HQ for all Time

Monte Carlo simulations were performed to evaluate the distribution of maximum ICRs and HQs for all time at the three compliance points. Results for each of the three compliance points are provided below.

Compliance Point C_1

The results for maximum ICRs beneath the trench are shown on Figure 4-2. Alternatives 1 and 2 have the highest ICRs, which reflects the absence of a liner and the high Sr-90 concentrations associated with the E waste. The Waste C vitrification alternative (Alternative 10) has the lowest ICR. In general, ICRs are lower for alternatives with liners than un-lined alternatives.

The four orders of magnitude range in the 90 percent certainty bounds for Alternatives 3, 4, and 5 reflect the high sensitivity of the results to Sr-90 travel time through the liner. The Sr-90 travel time through the liner was long enough in approximately 60 percent of the realizations that it decayed below the uranium risk level before it reached the bottom of the trench. Therefore, U-238 controlled the risk in those realizations (which accounted for the median risk of approximately 1.0E-03). In the remaining 40 percent of the realizations, Sr-90 controlled the risk. As a result, in a significant percentage of the simulations (approximately 5 percent) the risk exceeded 10.

The cancer risk associated with Sr-90 for the waste C alternatives is much less significant and therefore the results are less dependent upon Sr-90 travel times through the liner. This is reflected in the much smaller 90 percent certainty bounds for the waste C alternatives, and the lower risks in general.

The results for maximum HQ at C_1 are provided in Figure 4-3. Virtually no difference is apparent in the non-vitrification alternatives, reflecting the lack of dependence on travel time through the liner due to the zero decay assumption for Cr(VI). The waste C vitrification alternative (Alternative 10) has a significantly lower HQ, due to the dependence on the rate of dissolution instead of solubility.

Compliance Point C_2

The results for maximum ICR at the water table are shown on Figure 4-4. The highest ICRs (on the order of 10^{-3}) are associated with the no-barrier alternatives (Alternatives 1, 6, 7, 13, and 16). Although Alternative 10 has no barrier, the concentration is reduced due to vitrification. The alternatives with either a Hanford or RCRA barrier have very similar ICR distributions and tend to be an order of magnitude less than the no-barrier alternatives.

The uncertainty bounds for Alternatives 3, 4, and 5 are much smaller at C_2 compared with C_1. This reflects the lack of dependence on Sr-90, which is always decayed below significant risk levels before it reaches the water table. U-238 was the controlling contaminant for ICRs in all the realizations because it is the only carcinogen that is not significantly decayed before it reaches the water table.

The results for maximum HQ at C_2 are provided in Figure 4-5. The results are parallel to the ICR results, which reflects the fact that similar to U-238, Cr(VI) is assumed not to decay.

Compliance Point C_3

ICR and HQ results for C_3 are similar to those for C_2 except that risks are lowered due to dilution. The results for maximum ICR at the ERDF boundary are shown on Figure 4-6. The highest ICRs (on the order of 10^{-4} to 10^{-3}) are associated with the no-barrier alternatives (Alternatives 1, 6, 7, 13, and 16). The alternatives with either a Hanford or RCRA barrier have very similar ICR distributions that tend to be an order of magnitude lower than the no-barrier alternatives. HQ results, shown in Figure 4-7, reflect similar reductions (due to dilution in the saturated zone) as the ICR results.

The deterministic worst-case results, shown on Figure 4-6, were calculated using either the upper bound or the lower bound for the triangular distribution (whichever was conservative in terms of risk). These bounds correspond to either the zero percentile or the 100th percentile of the input parameters and therefore should provide the upper risk bound. In general, these worst-case estimates are between a factor of 7 (Alternative 6) and a factor of 90 (Alternative 14) greater than the 95th percentile determined using the probabilistic analysis. Since the probability of such risks occurring is zero, these worst case deterministic results are essentially meaningless.

4.2.2 Travel Times

Monte Carlo simulations were performed to evaluate U-238 travel times to the ERDF boundary. Only U-238 was addressed because it is the only significant cancer-causing risk driver at the ERDF boundary. The deterministic travel times, the median travel times, and the 90 percent certainty ranges for the alternatives are shown on Figure 4-8. It is clear from the figure that the no-barrier alternatives have significantly lower travel times (on the order of 10^4 years) than the barrier alternatives (10^5 years). The travel time differences between the modified RCRA barrier alternatives and the Hanford Barrier alternatives are virtually indistinguishable.

The same pattern is evident in the Cr(VI) travel times, shown in Figure 4-9. The Cr(VI) travel times to the ERDF boundary are on the order of 100 to 1,000 years for the no-barrier alternatives and on the order of 2,000 to 20,000 years for the barrier alternatives.

4.2.3 Maximum ICR and HQ within 10,000 Years

Due to the uncertainties regarding facility performance far into the future, the design criteria may be based on performance within a certain time limit. The deterministic results indicated that the maximum ICR within 10,000 years at the water table and at the ERDF boundary was zero. To estimate the uncertainty of these results, a Monte Carlo analysis was performed to estimate risk at the ERDF boundary within 10,000 years.

Results for maximum ICRs at the ERDF boundary within 10,000 years are shown on Figure 4-10. The median ICR is less than 1.0E-09 for all the alternatives. The 90 percent certainty bounds only extend above 1.0E-09 for alternatives without barriers (Alternatives 1, 6, 7, 10, 13, and 16). The upper certainty bounds corresponds to the 95 percent certainty value. For example, there is 95 percent certainty that the ICR is less than 1.0E-03 for Alternative 1 (i.e., 5 percent of the realization exceeded 1.0E-03). These results indicate that the vitrification without a barrier alternatives (Alternatives 6 and 10) are less protective than the alternatives with a barrier.

The results for maximum HQ at the ERDF boundary within 10,000 years are shown in Figure 4-11. The median and upper bounds are not significantly changed from those presented for all time. However, the lower certainty bounds for the barrier alternatives are equal to zero. This is because Cr(VI) did not reach the ERDF boundary within 10,000 years in approximately 30 percent of the realizations.

4.2.4 Importance of Selecting the Comparison Criteria

As illustrated above, the performance scores for each alternative will depend on the criteria selected. Given the results presented above, examples of performance criteria include:

- Median maximum risk and hazard quotient with no time constraint,
- 95th percentile of maximum risk and hazard quotient,
- Median travel time of U-238 and Cr(VI),
- 95th percentile of maximum risk and hazard quotient within 10,000 years,
- Probability that risk will be less than a certain risk level (i.e. 10^{-5}) and hazard quotient less than a certain HQ level (i.e. 1).

One of these criteria may be identified as the single long-term effectiveness measure. Alternatively, multiple measures could be subcriteria within the overall long-term effectiveness criteria with different weights for each measure. Examples of the relative scores are provided in Tables 4-11 and 4-12 for different criteria. It is clear that different criteria will result in differences in both rank and the relative magnitude of difference. For the probabilistic measures, alternatives without a barrier perform the worst and alternatives with a Hanford barrier always perform the best.

4.2.5 Relative Performance

As illustrated in Figure 4-6, comparison of result distributions may not indicate distinct performance differences. Use of relative performance information can illustrate consistent performance differences that may not be apparent when results for alternatives are viewed independently. In addition, the certainty that one alternative will perform better than another can be determined using relative performance information. Relative performance between alternatives can be determined by dividing the result for one alternative by the result of another alternative for each realization. If the ratio equals 1 then the alternatives perform the same. Deviations from a ratio of 1 reflect differences in performance. It is important that the ratio be calculated for each realization so that common parameters will be the same and the differences will reflect actual design differences. Examples of relative performance are illustrated in Figures 4-12 and 4-13.

The distribution of maximum risk ratios for Alternative 11 (Hanford barrier and a single liner) versus Alternative 9 (the modified RCRA barrier and a MTR liner) is provided in Figure 4-12. The 90 percent certainty range is 0.4 to 1.5 and the median value is 0.8 (indicating that the maximum risk for Alternative 11 is on average 20 percent lower than

Alternative 9). The certainty that the Hanford barrier performs better than the modified RCRA barrier is approximately 70 percent (measured by the probability that the maximum risk ratio is less than 1).

The distribution of maximum risk ratios for Alternative 11 (Hanford barrier and a single liner) versus Alternative 8 (Hanford barrier alone) is provided in Figure 4-13. The 100 percent certainty range is 0.9997 to 0.9999996, indicating that the performance of these alternatives is virtually identical but that Alternative 11 always performs slightly better than Alternative 8. This is consistent with the longer travel time for Alternative 11 (since the liner reduces the depth of the trench) and the subsequent reduction in U-238 concentration due to decay. The reduction in U-238 concentration is very minor due to its long half-life.

4.3 SENSITIVITY ANALYSES

A sensitivity analysis determines the effects of variation(s) in one or more parameters on the performance of a system or some part of it (Brandsetter and Buxton 1989). In addition, the performance measures are influenced more by some input parameters than others (Brandstetter and Buxton 1989). Sensitivity studies can help determine the relative importance of input parameters. Sensitivity analyses were performed using 1,500 realizations. The best estimates and ranges for the input parameters are provided in Table 3-1 through 3-4. Sensitivity is measured using rank correlation. Rank correlations are preferred over standard correlation coefficients because they are valid even if the relationship between the parameter and the result is nonlinear. Similar to standard correlation coefficients, rank correlations vary between -1 and +1, where zero signifies no correlation and +1 and -1 signify perfect correlations. Positive rank correlations indicate that the results generally increase as the parameter increases and negative rank correlations indicate that the results decrease as the parameter decreases. Correlation between the barrier infiltration rates and the infiltration rate between the trenches was set equal to zero to avoid interference between these parameters.

The same variables and distributions of the variables used for the uncertainty analysis were used for the sensitivity analysis. The sensitivity analysis only addresses Waste C alternatives, although the conclusions should be similar for Waste E and Waste A alternatives. Sensitivity analyses were performed for the following criteria:

- Maximum risk for all time,
- Travel time to the ERDF boundary,
- Maximum risk within 10,000 years,
- Relative maximum risk.

4.3.1 Maximum Risk for all Time

The rank correlations for Alternative 7 (no action) with regards to the maximum risk at the ERDF boundary are provided in Figure 4-14. The important parameters (in order by rank) are U-238 solubility, the no-barrier infiltration rate, the saturated zone mixing depth, saturated zone hydraulic conductivity and hydraulic gradient, and the vadose zone mixing factor. The remaining parameters are not significantly correlated with maximum risk. Alternative 16 has the same sensitivity patterns as Alternative 7.

The rank correlations for Alternative 8 (Hanford barrier) with regards to the maximum risk at the ERDF boundary are provided in Figure 4-15. The important parameters (in order by rank) are U-238 solubility, the vadose zone mixing factor, the Hanford barrier infiltration rate, the saturated zone hydraulic conductivity, the mixing depth, and the saturated zone hydraulic gradient. In comparison with the no-action Alternative 7, the vadose zone mixing factor is more important due to the reduced infiltration rate through the barrier. Alternatives 9, 11, 12, and 15 have the same sensitivity patterns as Alternative 8.

The rank correlations for Alternative 10 (vitrification) with regards to the maximum risk at the ERDF boundary are provided in Figure 4-16. The important parameters (in order by rank) are the partitioning coefficient for U-238, the vitrified material dissolution rate, the saturated zone hydraulic conductivity and mixing depth, and the vadose zone mixing factor. The addition of the partitioning coefficient for U-238 and the vitrified material dissolution rate to the list of significant parameters is due to the fact that the dissolution of vitrified material, (which is a function of the partitioning coefficient and the dissolution rate), controls the initial leachate concentration (C_0). The elimination of U-238 solubility from the list of significant parameters is because solubility does not control the initial leachate concentration.

4.3.2 Travel Time to the ERDF Boundary

Figure 4-17 shows the rank correlations for Alternative 7 (no action) with regards to U-238 travel time to the ERDF boundary. The important parameters (in order by rank) are the partitioning coefficient for U-238 and the no-barrier infiltration rate. The remaining parameters are not significantly correlated with U-238 travel time.

The rank correlations for Alternative 8 (Hanford barrier) with regards to U-238 travel time to the ERDF boundary are provided in Figure 4-18. The important parameters (in order by rank) are the partitioning coefficient for U-238, the Hanford barrier infiltration rate, and the vadose zone mixing depth. The vadose zone mixing depth becomes influential for Alternative 8 because the presence of a barrier creates a significant contrast between the infiltration rate through the trench and the infiltration rate between the trenches. Alternatives without a barrier (10 and 16) have the same sensitivity patterns as Alternative 7 and alternatives with a barrier (9, 11, 12, and 15) have the same sensitivity patterns as Alternative 8.

4.3.3 Maximum Risk within 10,000 Years

The rank correlations for Alternative 7 (no action) with regards to the risk within 10,000 years at the ERDF boundary are provided in Figure 4-19. The important parameters (in order by rank) are the no-barrier infiltration rate and the partitioning coefficient for U-238.

The rank correlations for Alternative 8 (Hanford barrier) with regards to the risk at 10,000 years at the ERDF boundary are provided in Figure 4-20. None of the input parameters has a significant correlation with the risk within 10,000 years at the ERDF boundary. This is because contaminants do not reach the ERDF boundary within 10,000 years in all but a few of the realizations. Therefore, alternatives without a barrier (10 and 16) have the same sensitivity patterns as Alternative 7 and alternatives with a barrier (9, 11, 12, and 15) have the same sensitivity patterns as Alternative 8.

4.3.4 Relative Maximum Risk for all Time

Sensitivity analyses for relative performance were also conducted. The examples discussed below address important parameters for relative performance between alternatives with similar performance but distinct design.

The rank correlations of maximum risk ratios for Alternative 9 versus Alternative 8 are shown in Figure 4-21. The important parameters are the Hanford barrier infiltration rate (Alternative 8) and the RCRA barrier infiltration rate (Alternative 9). The remaining parameters are not important because the barrier type is the only difference between these alternatives.

The rank correlations of maximum risk ratios for Alternative 10 versus Alternative 8 are presented in Figure 4-22. The important parameters (in order by rank) are U-238 solubility, the partitioning coefficient for U-238, the Hanford barrier infiltration rate, the vitrified material dissolution rate, and the vadose zone mixing factor. As described above, U-238 solubility is the controlling parameter for the initial leachate concentrations of Alternative 2, while the partitioning coefficient and the dissolution of vitrified material control the initial leachate concentrations of Alternative 10. Because the controlling mechanisms for the initial leachate concentrations are different for Alternative 8 and 10, the relative performance between these two alternatives are sensitive to the parameters that govern these mechanisms. In addition, Hanford barrier infiltration rate and vadose zone mixing factor are important because of the difference in barriers (Alternative 8 has no barrier and Alternative 10 has a Hanford barrier).

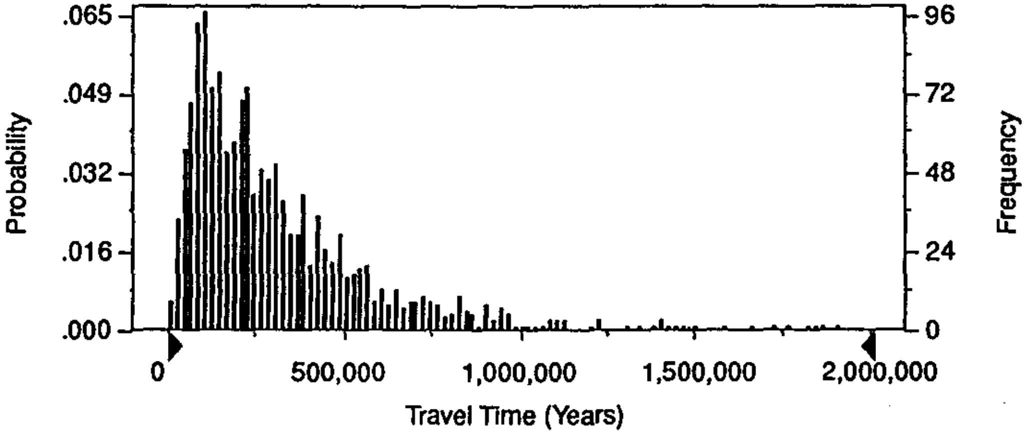
4.3.5 Summary of Sensitivity Analyses

In general, the most important parameters for maximum risk (without time constraints) are U-238 solubility, the infiltration rate through the barrier, the saturated zone hydraulic conductivity and hydraulic gradient, the vadose zone mixing factor (for alternatives with a infiltration barrier), and the dissolution rate of vitrified material and the partitioning coefficient for U-238 (for alternative with vitrification).

The risk within 10,000 years for no-barrier alternatives is most sensitive to the infiltration rate and the partitioning coefficient of U-238 (for no-barrier alternatives). For alternatives with an infiltration barrier, there is no correlation between the risk within 10,000 years and the input parameters due to the fact that the travel time is almost always greater than 10,000 years.

The important parameters regarding travel time to the ERDF boundary are the partitioning coefficient, infiltration rate, and the vadose zone mixing depth (for alternatives with a barrier).

For relative risk comparisons between two alternatives, the important parameters are a subset of the parameters that control the risk estimates for the individual alternatives. Only those parameters that affect design differences between the alternatives are included in the subset.



Percentiles for Entire Range:

Percentile	Travel Time (Years)
0%	6,774
5%	52,978
25%	127,898
50%	236,547
75%	433,972
95%	937,260
100%	13,714,823

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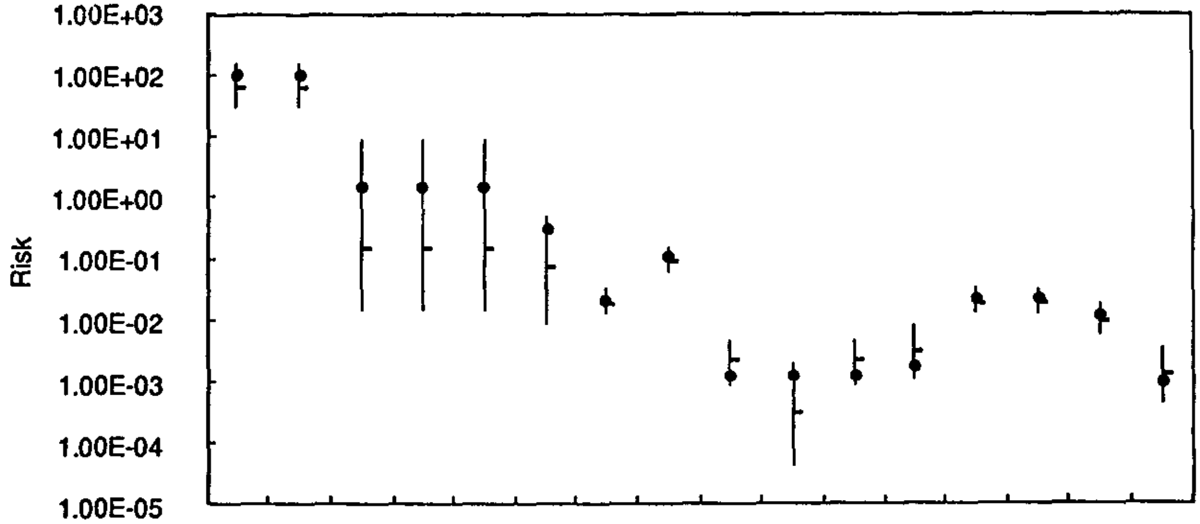
Figure 4-1. Frequency Chart for Alternative 8 - Uranium-238 Travel Time to the ERDF Boundary.



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16	13	14
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C	A	A
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW	NW	W
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F	NT	V
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL	NL	V
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB	NB	R

LEGEND

95th percentile
Median value
Deterministic
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

Treatment
NT – No Treatment
F – Fixation
V – Vitrification

Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

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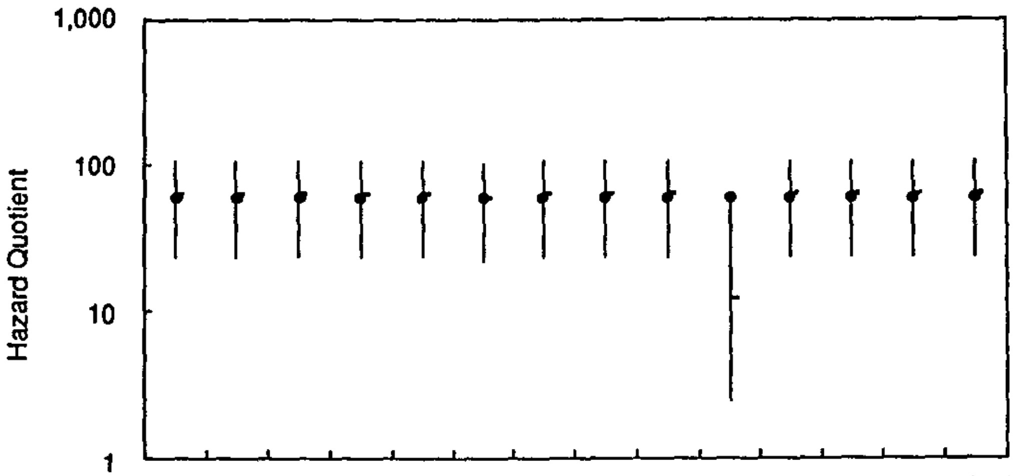
Figure 4-2. Maximum Risk Beneath the Trench.



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB

LEGEND

95th percentile
Median value
Deterministic
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

Treatment
NT – No Treatment
F – Fixation
V – Vitrification

Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

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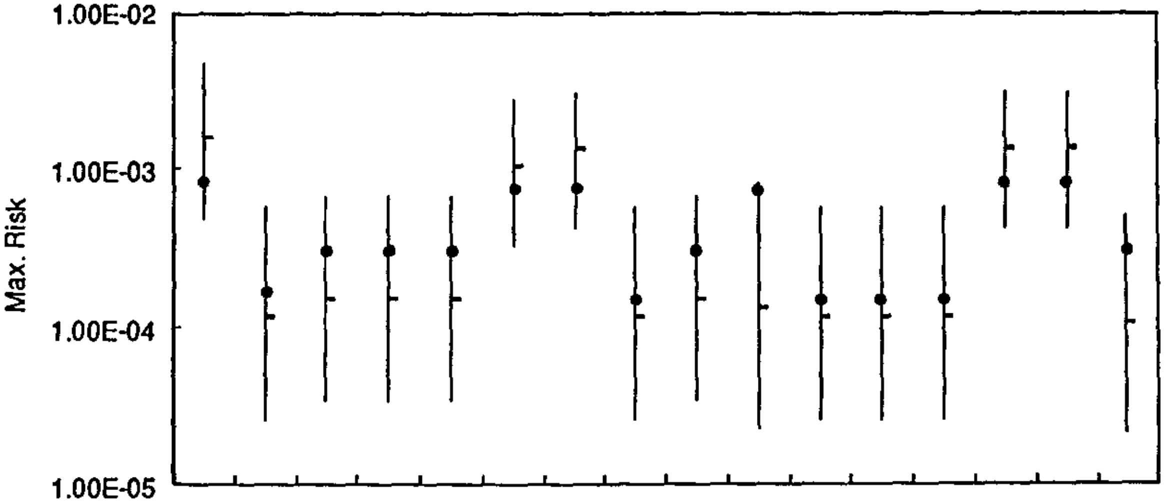
Figure 4-3. Hazard Quotient of Chromium(VI) Beneath the Trench.



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16	13	14
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C	A	A
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW	NW	W
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F	NT	V
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL	NL	V
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB	NB	R

LEGEND
95th percentile
Median value
Deterministic
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

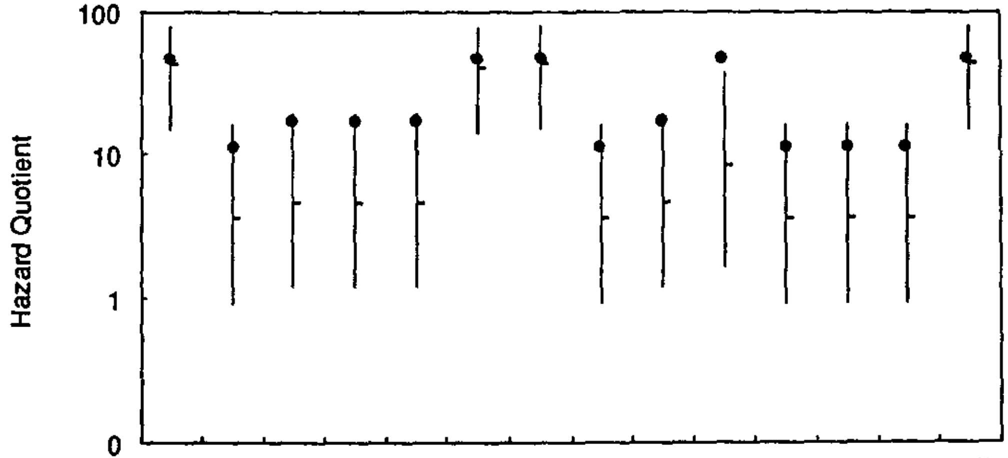
Treatment
NT – No Treatment
F – Fixation
V – Vitrification

Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46264/9-20-93

Figure 4-4. Maximum Risk at the Water Table.



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB

LEGEND

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

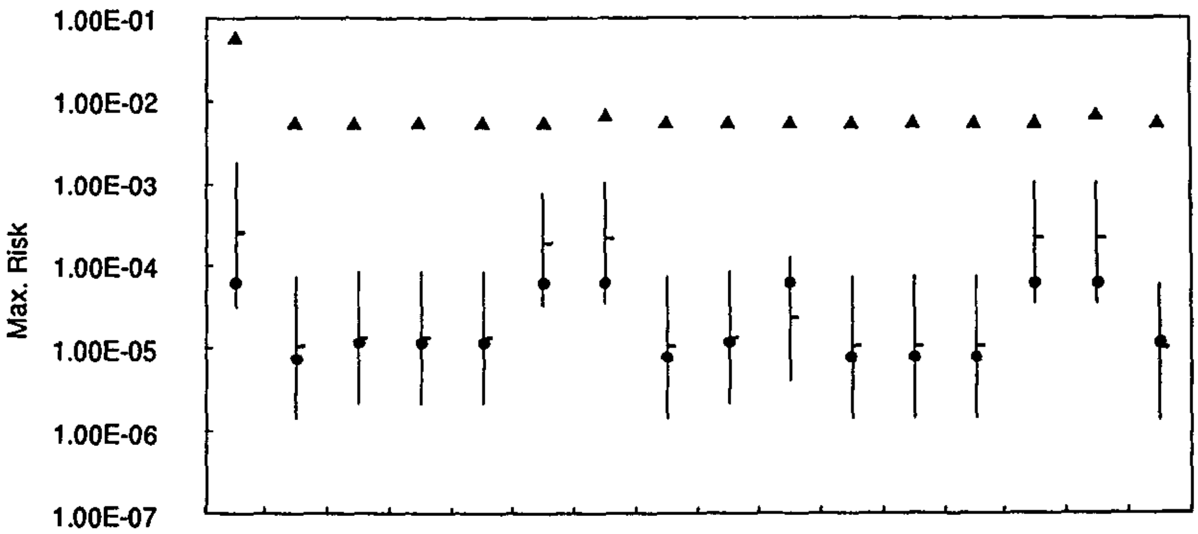
Treatment
NT – No Treatment
F – Fixation
V – Vitrification

Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46267/9-20-93

Figure 4-5. Hazard Quotient of Chromium(VI) at the Water Table.



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16	13	14
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C	A	A
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW	NW	W
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F	NT	V
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL	NL	V
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB	NB	R

LEGEND
▲ Deterministic worst case
95th percentile
Median value
● Deterministic
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

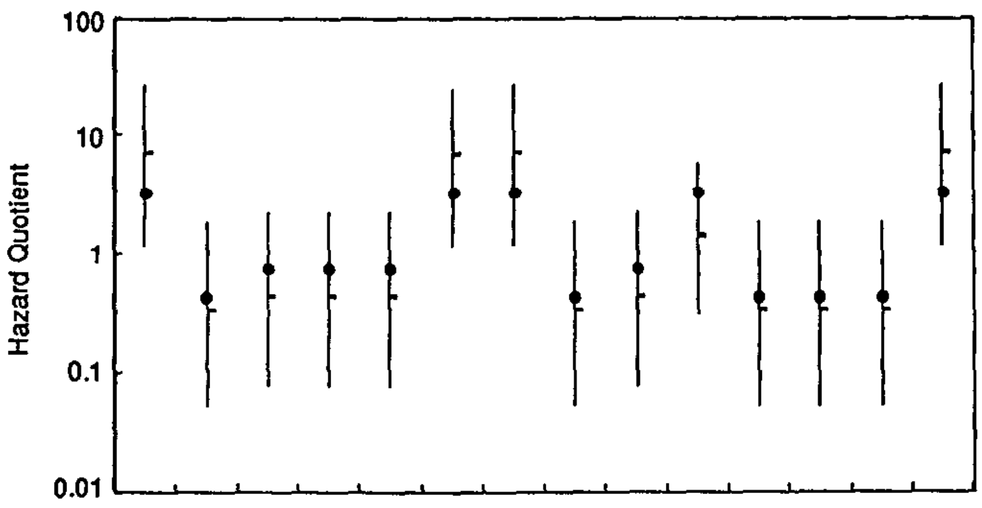
Treatment
NT – No Treatment
F – Fixation
V – Vitrification

Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46265/9-20-93

Figure 4-6. Maximum Risk at the ERDF Boundary.



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB

LEGEND
- 95th percentile
- Median value
- Deterministic
- 5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

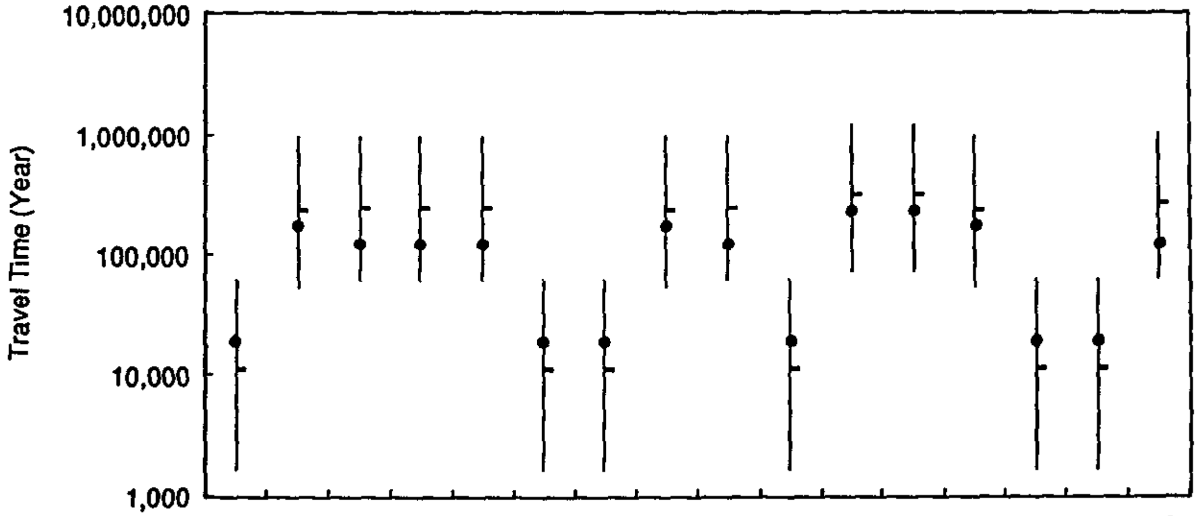
Treatment
NT – No Treatment
F – Fixation
V – Vitrification

Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46268/9-20-93

Figure 4-7. Hazard Quotient of Chromium(VI) at the ERDF Boundary.



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16	13	14
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C	A	A
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW	NW	W
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F	NT	V
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL	NL	V
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB	NB	R

LEGEND

95th percentile
Median value
Deterministic
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

Treatment
NT – No Treatment
F – Fixation
V – Vitrification

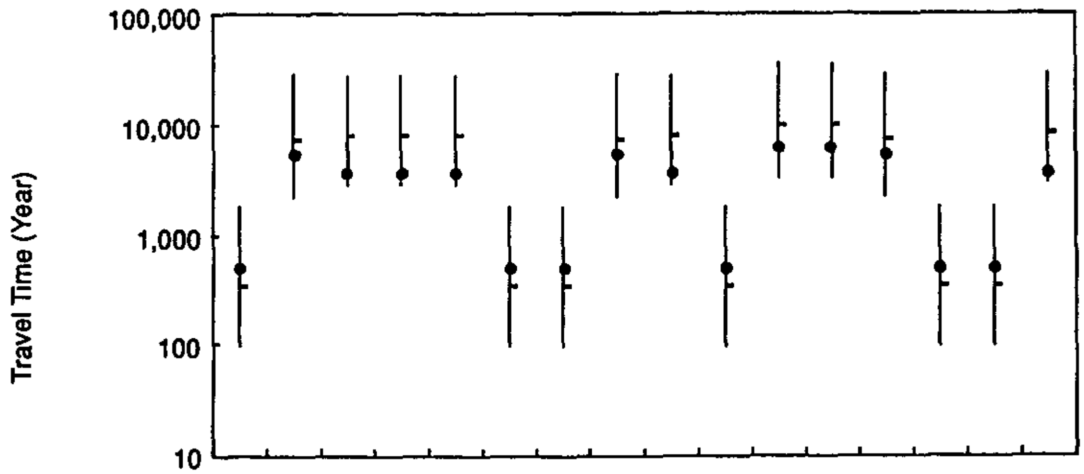
Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46261/9-20-93

Figure 4-8. Travel Time of Uranium-238 to the ERDF Boundary.

9413280.0246



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	A	A
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	W
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	NT	V
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	V
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	NB	R

LEGEND

95th percentile
Median value
Deterministic
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

Treatment
NT – No Treatment
F – Fixation
V – Vitrification

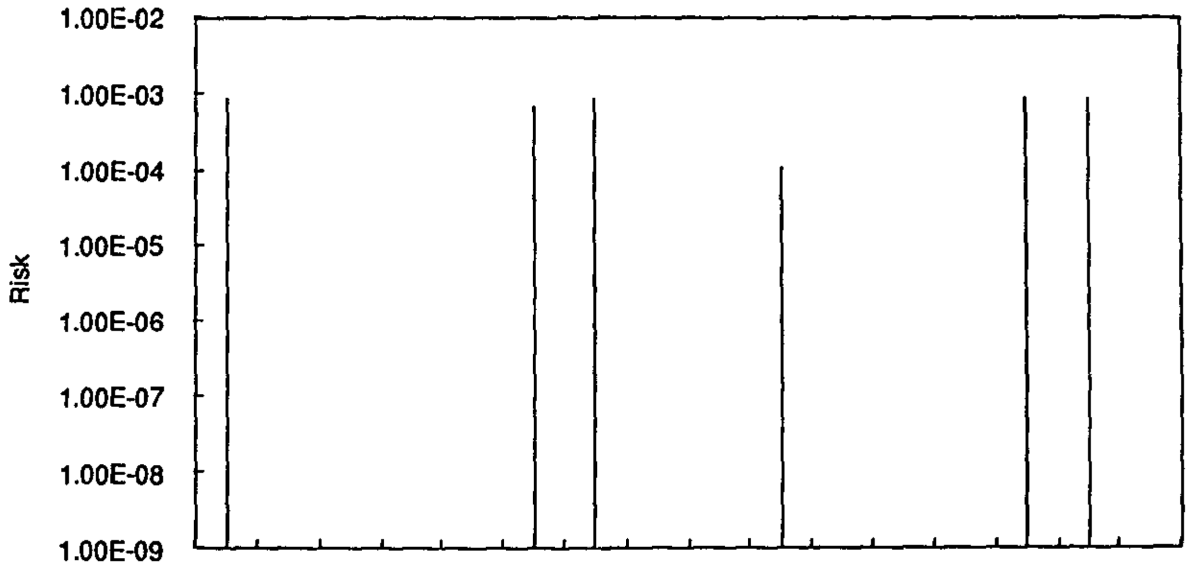
Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46262/9-29-93

Figure 4-9. Travel Time of Chromium(VI) to the ERDF Boundary.

9413280.0247



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16	13	14
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C	A	A
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW	NW	W
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F	NT	V
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL	NL	V
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB	NB	R

LEGEND

95th percentile
Median value
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

Treatment
NT – No Treatment
F – Fixation
V – Vitrification

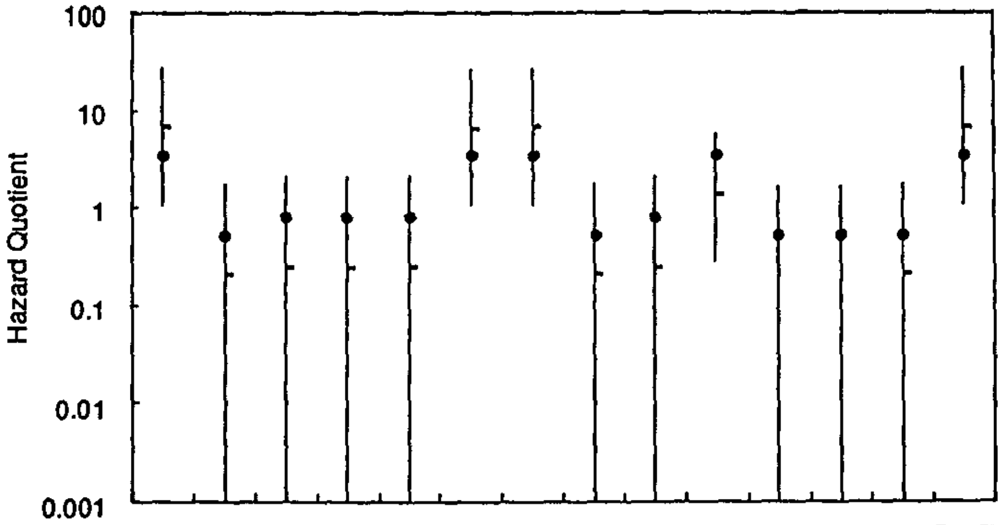
Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46269/9-20-93

Figure 4-10. Risk Within 10,000 Years at the ERDF Boundary.

9413280.0248



Alternative	1	2	3	4	5	6	7	8	9	10	11	12	15	16
Waste Type	E	E	E	E	E	E	C	C	C	C	C	C	C	C
Soil Washing	NW	NW	NW	NW	NW	NW	NW	W	NW	W	NW	W	NW	NW
Treatment	NT	NT	NT	NT	F	V	NT	NT	F	V	NT	F	F	F
Liner	NL	NL	DL	SL	DL	NL	NL	NL	DL	NL	SL	SL	NL	NL
Barrier	NB	H	R	R	R	NB	NB	H	R	NB	H	H	H	NB

LEGEND

95th percentile
Median value
Deterministic
5th percentile

Waste Type
E – E Waste
C – C Waste
A – A Waste

Soil Washing
W – Soil Washing
NW – No Soil Washing

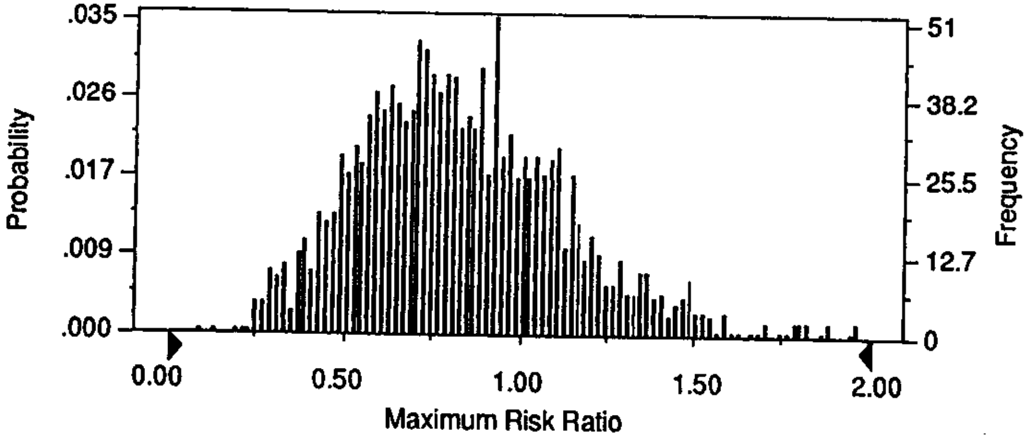
Treatment
NT – No Treatment
F – Fixation
V – Vitrification

Liner
NL – No Liner
SL – Single Liner
DL – MTR Liner
V – Vault

Barrier
NB – No Barrier
H – Hanford Barrier
R – RCRA Barrier

923 E034/46270/9-20-93

Figure 4-11. Hazard Quotient of Chromium(VI) Within 10,000 Years at the ERDF Boundary.

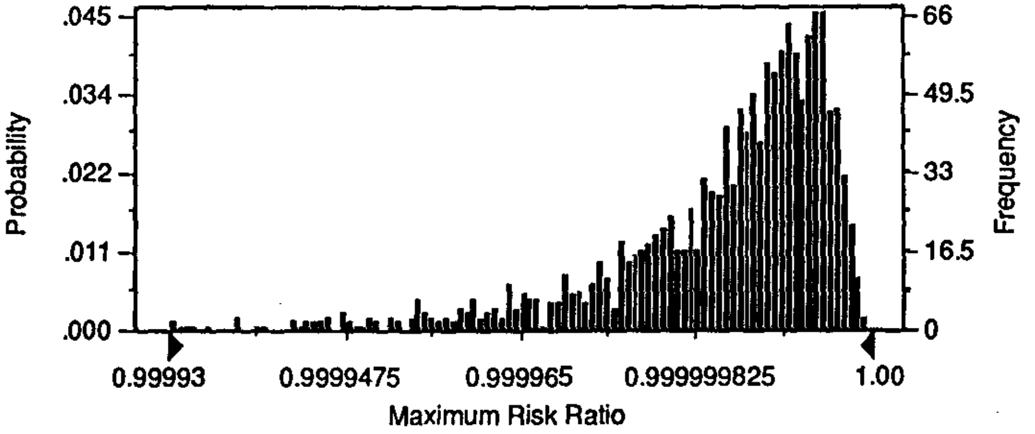


Percentiles for Entire Range:

Percentile	Maximum Risk Ratio
0%	0.08
5%	0.40
25%	0.62
50%	0.81
75%	1.04
95%	1.46
100%	4.90

923 E034/46035/9-20-93

Figure 4-12. Distributions of Maximum Risk Ratios at the ERDF Boundary for Alternative 11 Versus Alternative 9.

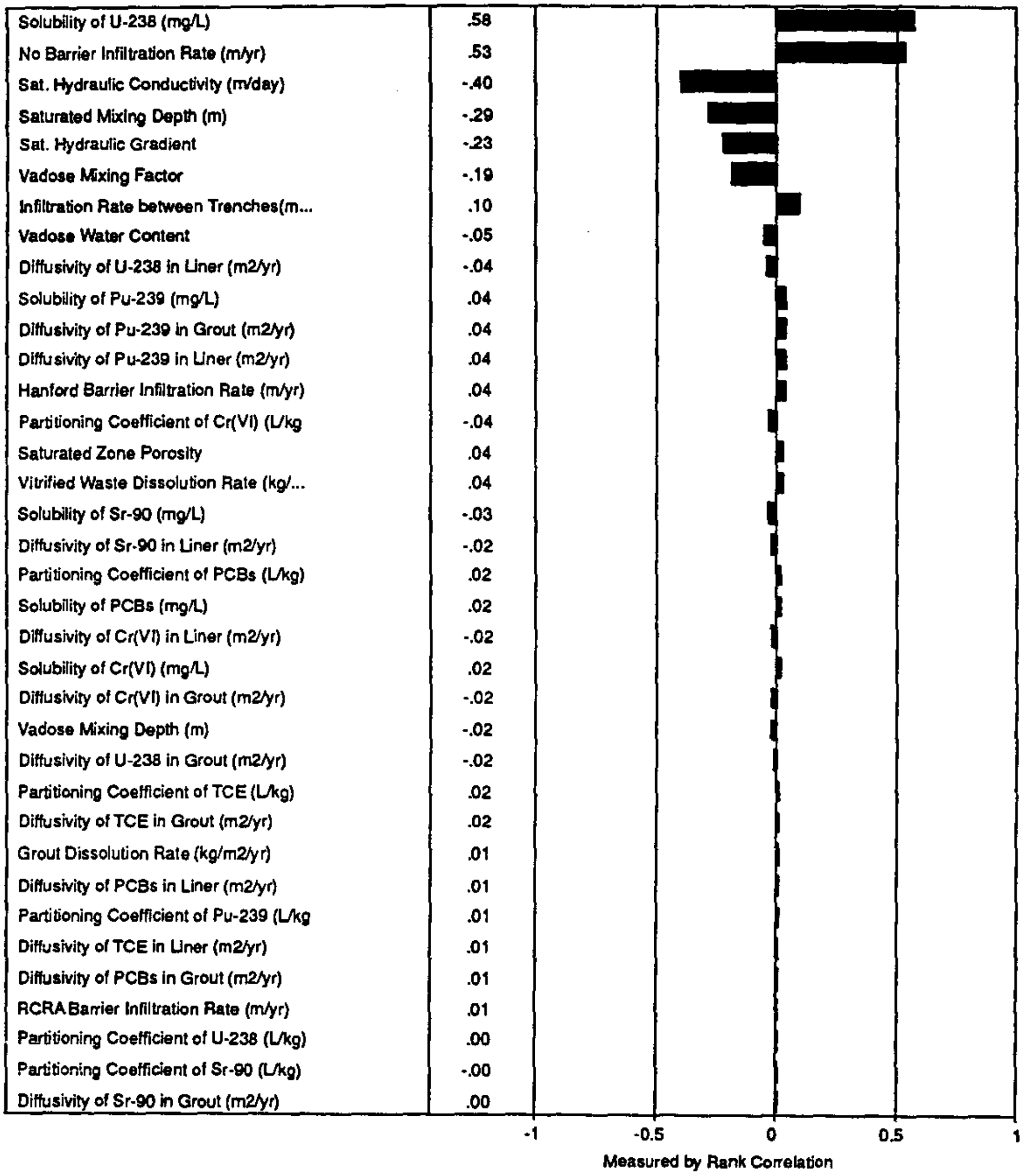


Percentiles for Entire Range:

Percentile	Maximum Risk Ratio
0%	0.999743677
5%	0.9999545
25%	0.999979557
50%	0.999988447
75%	0.999993366
95%	0.999997142
100%	0.999999622

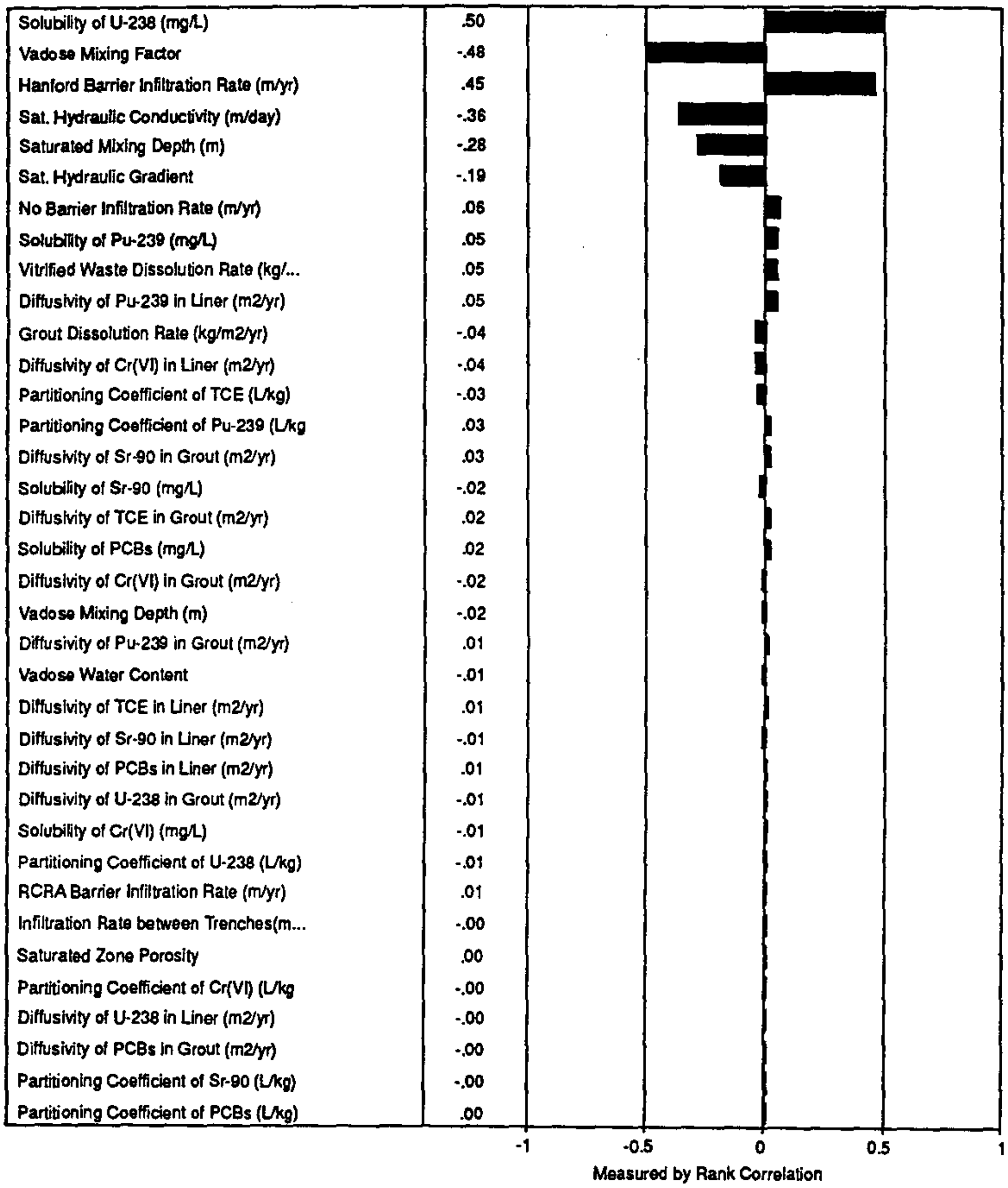
923 E034/46036/9-20-93

Figure 4-13. Distributions of Maximum Risk Ratios at the ERDF Boundary for Alternative 11 Versus Alternative 8.



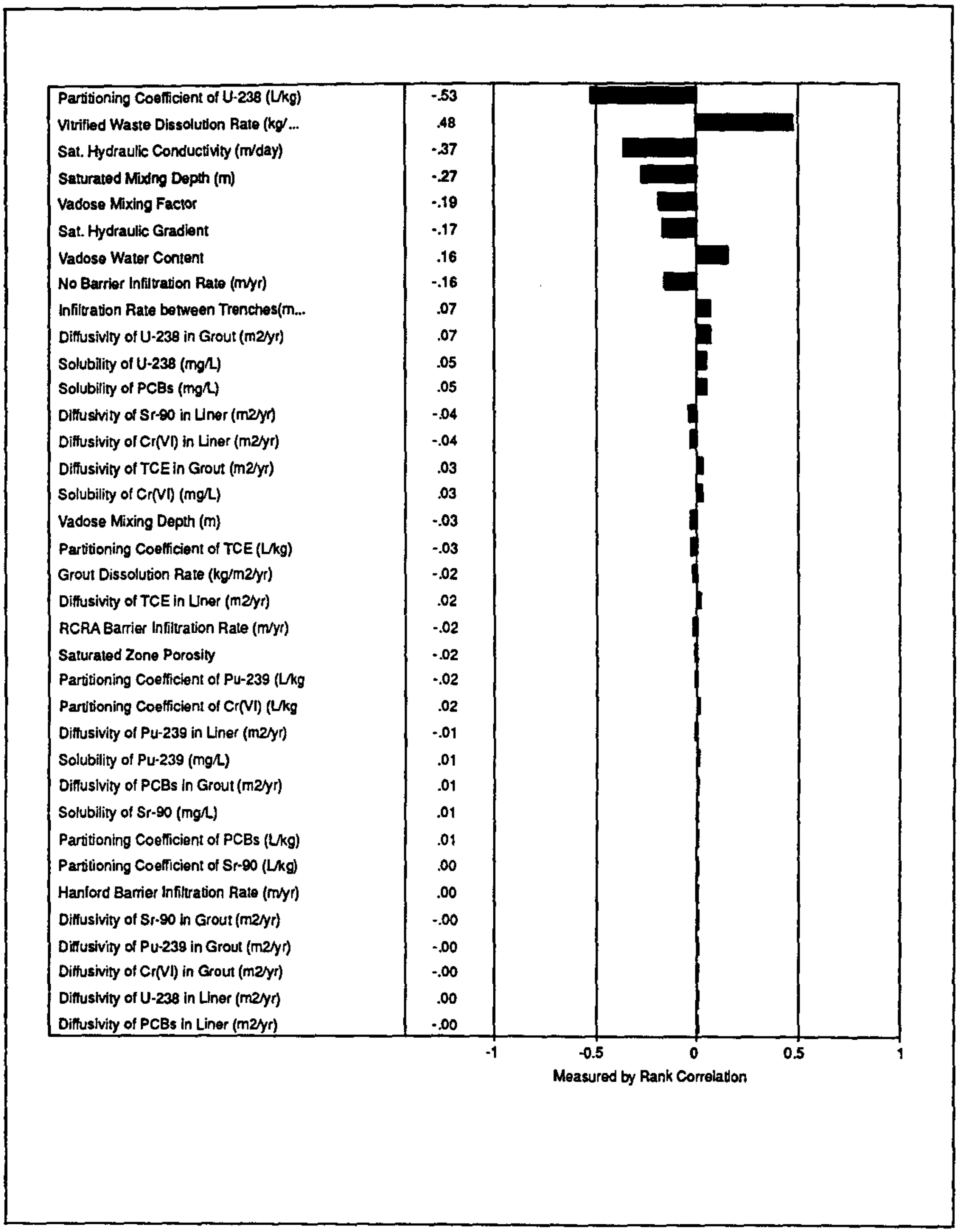
923-E034/46037/9-20-93

Figure 4-14. The Rank Correlations for Alternative 7 with Regards to the Maximum Risk at the ERDF Boundary.



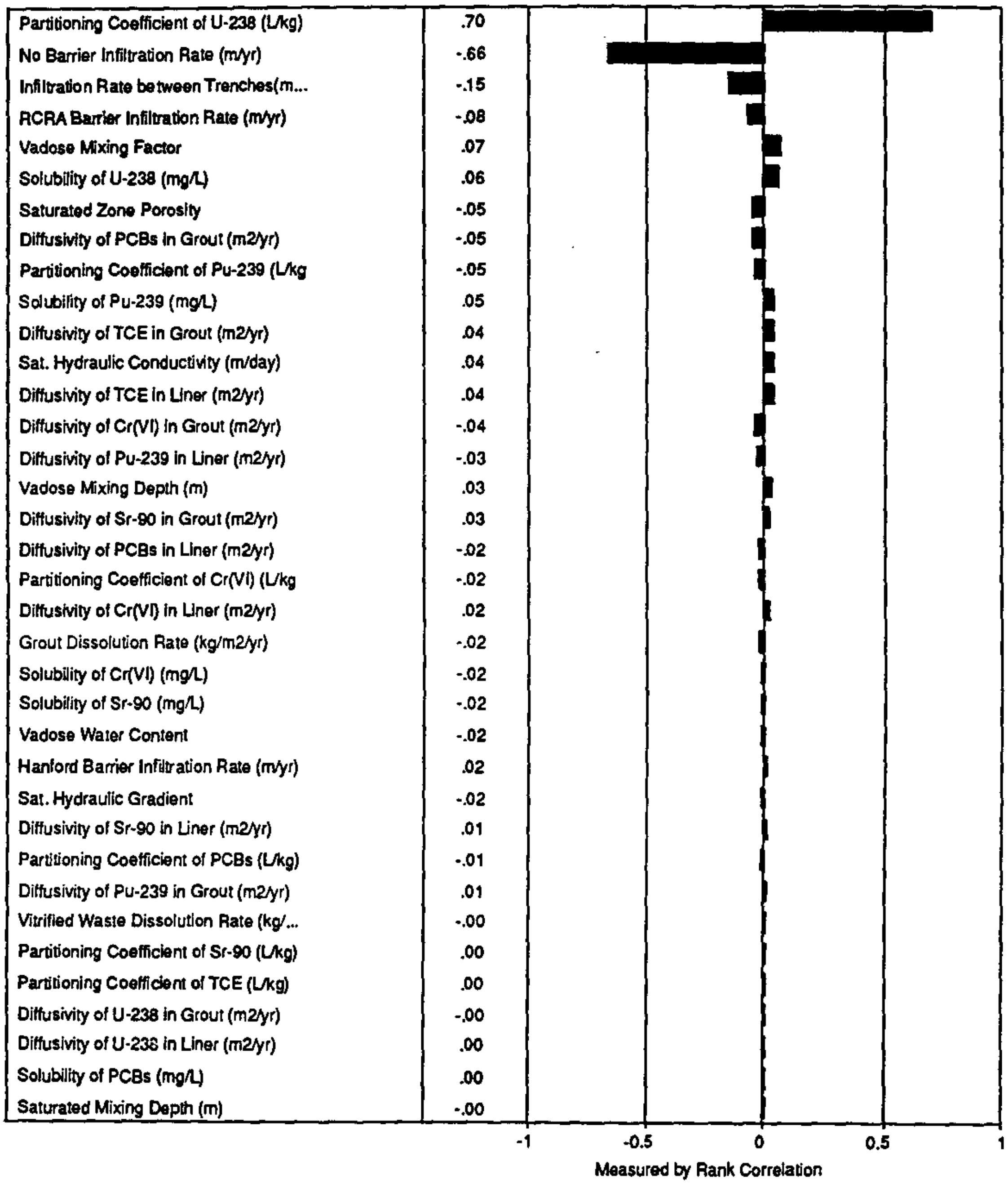
923-E034/46038/9-20-93

Figure 4-15. The Rank Correlations for Alternative 8 with Regards to the Maximum Risk at the ERDF Boundary.



923-E034/46039/9-20-93

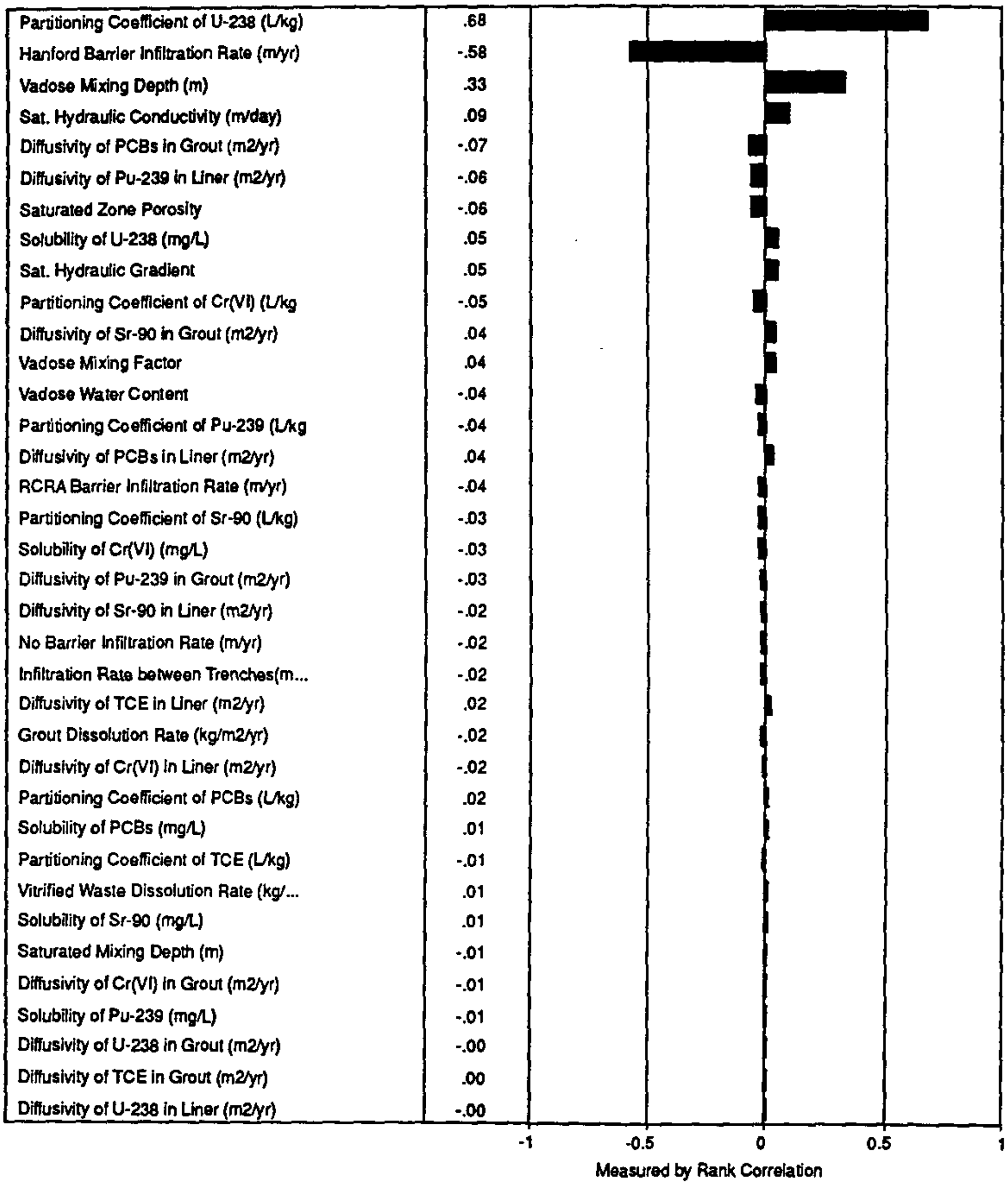
Figure 4-16. The Rank Correlations for Alternative 10 with Regards to the Maximum Risk at the ERDF Boundary.



923-E034/46040/9-20-93

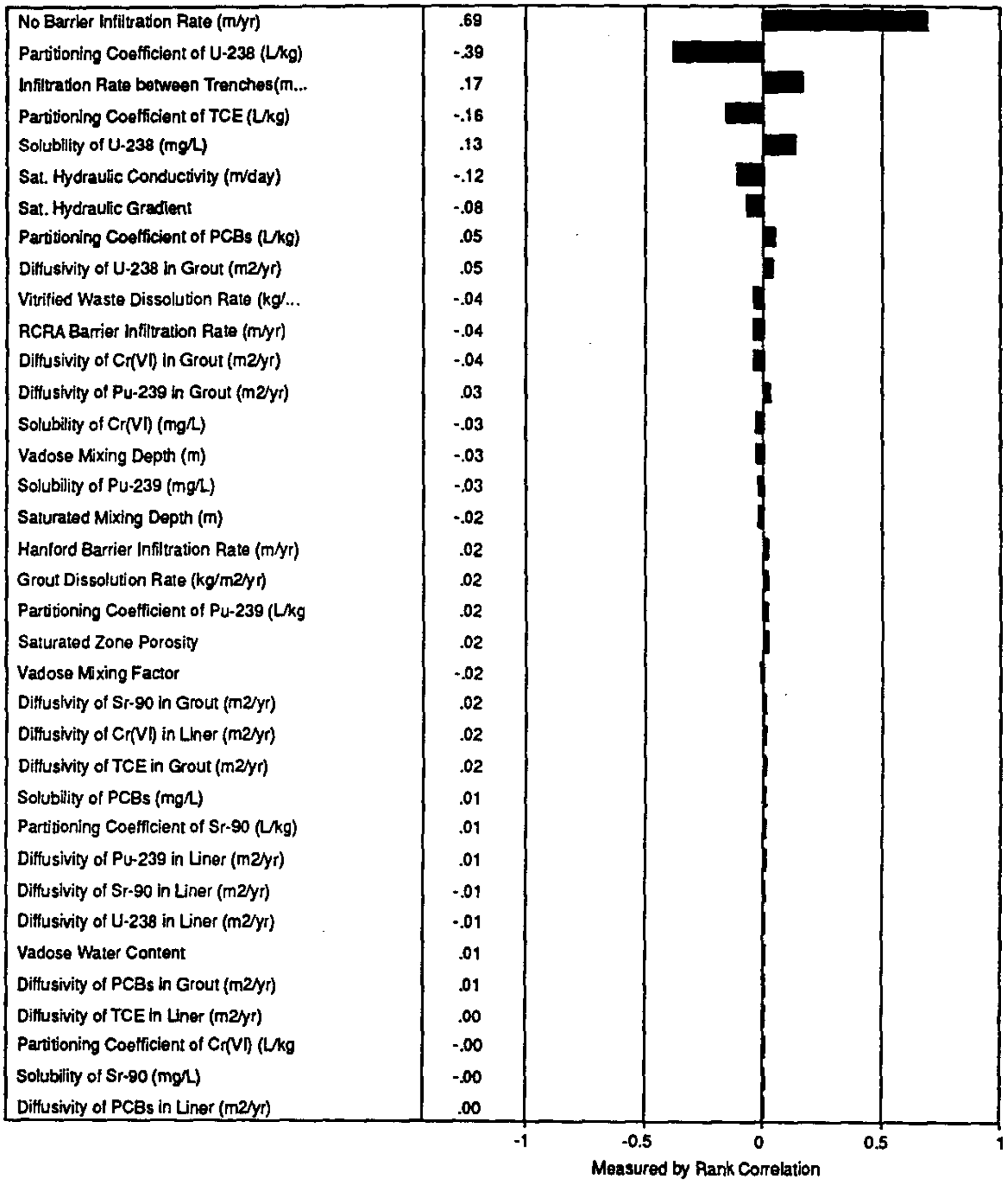
Figure 4-17. The Rank Correlations for Alternative 7 with Regards to the Travel Time of U-238 to the ERDF Boundary.

9413280.0255



923-E034/46041/9-20-93

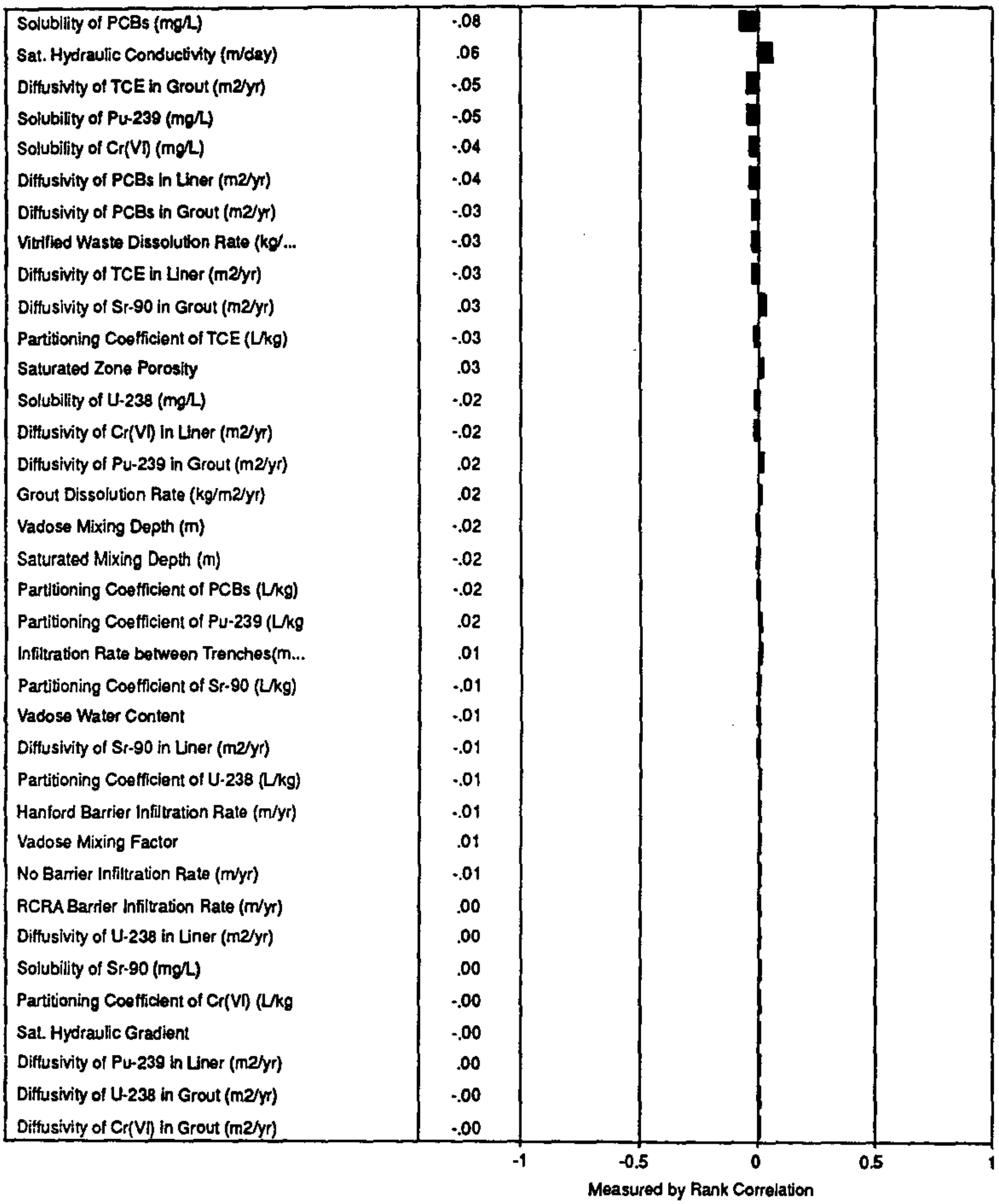
Figure 4-18. The Rank Correlations for Alternative 8 with Regards to the Travel Time of U-238 to the ERDF Boundary.



923-E034/46042/9-20-93

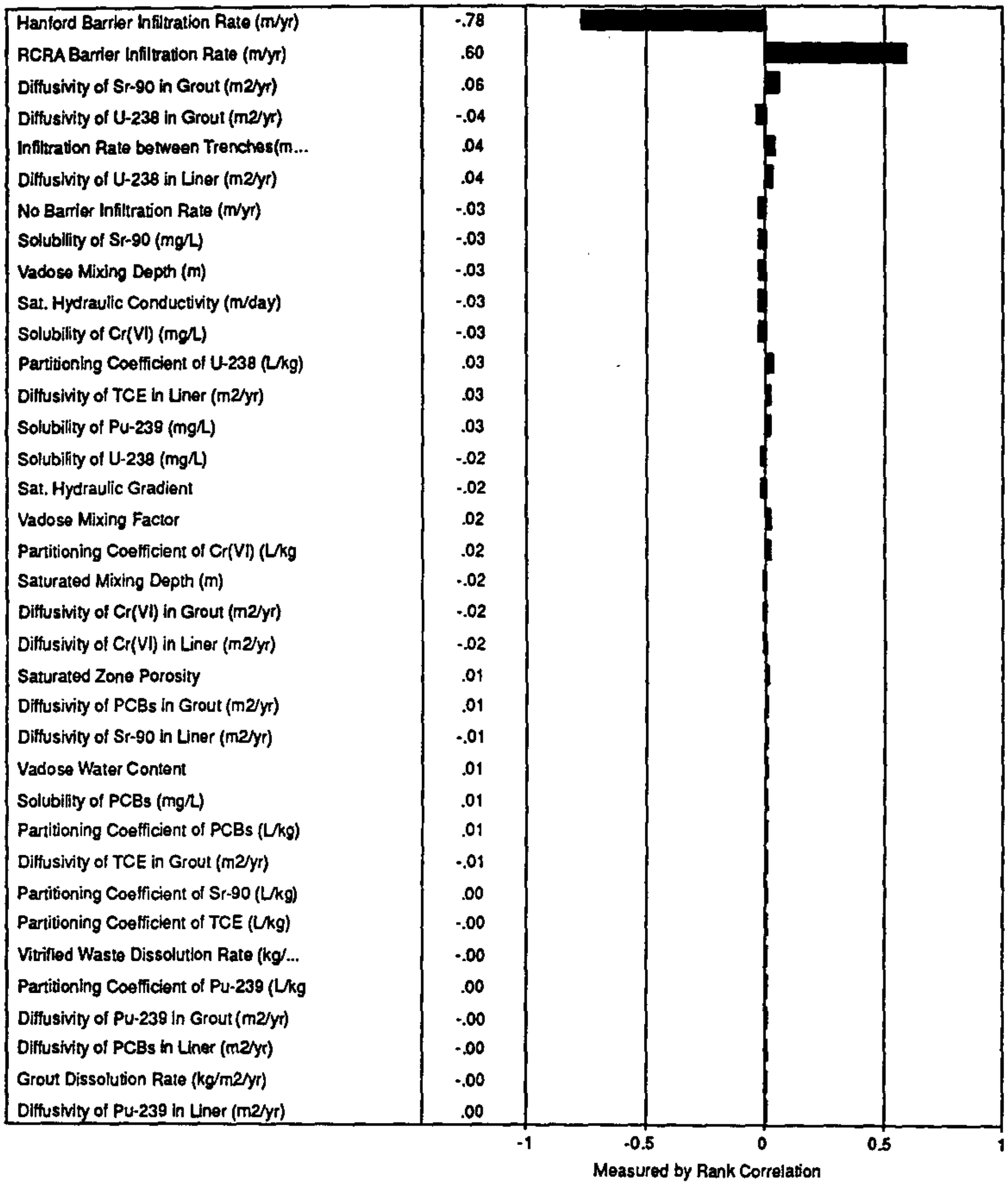
Figure 4-19. The Rank Correlations for Alternative 7 with Regards to the Risk Within 10,000 Years at the ERDF Boundary.

9413280.0257



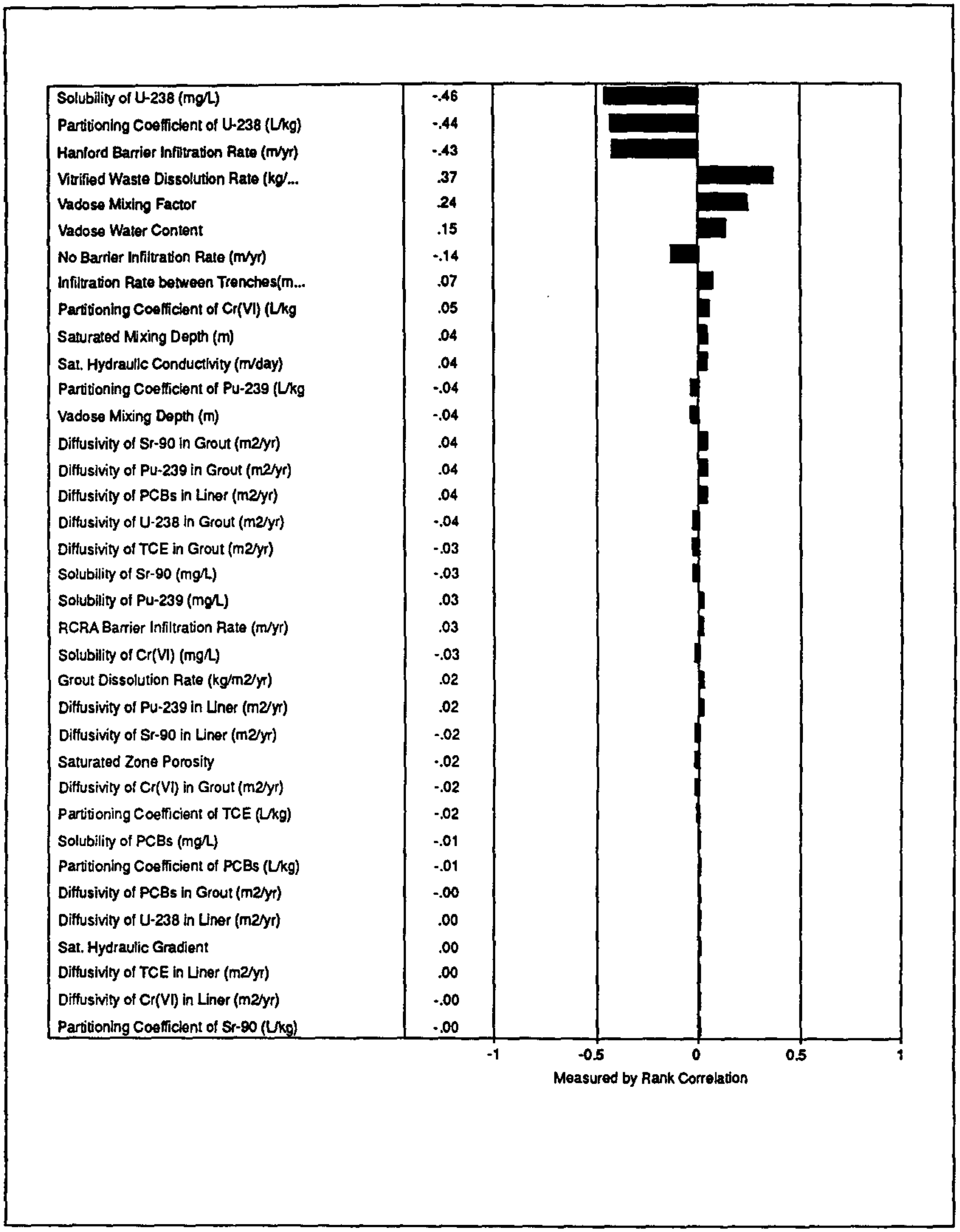
923-E034/46043/9-20-93

Figure 4-20. The Rank Correlations for Alternative 8 with Regards to the Risk Within 10,000 Years at the ERDF Boundary.



923-E034/46044/9-20-93

Figure 4-21. The Rank Correlations of the Maximum Risk Ratios for Alternative 9 Versus Alternative 8.



923-E034/46045/9-20-93

Figure 4-22. The Rank Correlations of the Maximum Risk Ratios for Alternative 10 Versus Alternative 8.

Table 4-1. Best Estimate Initial Leachate Concentrations (C_0) for Waste E Alternatives.

Alternative	1	2	3	4	5	6
Treatment	No Treatment	No Treatment	No Treatment	No Treatment	Fixation	Vitrification
Barrier	No Barrier	Hanford	RCRA	RCRA	RCRA	No Barrier
Liner	No Liner	No Liner	MTR	Single Liner	MTR	No Liner
Volume Reduction	1.00	1.00	1.00	1.00	1.00	1.00
Constituent	Initial Leachate Concentrations					
U-238 (pCi/L)	1.68E+03	1.68E+03	1.68E+03	1.68E+03	1.68E+03	1.68E+03
Sr-90 (pCi/L)	1.25E+08	1.25E+08	1.25E+08	1.25E+08	1.25E+08	4.09E+05
Pu-239 (pCi/L)	2.00E+04	2.00E+04	2.00E+04	2.00E+04	2.00E+04	6.58E+01
Cr (VI) (mg/L)	5.00E+00	5.00E+00	5.00E+00	5.00E+00	5.00E+00	5.00E+00
TCE (mg/L)	1.89E+02	1.89E+02	1.89E+02	1.89E+02	1.89E+02	6.22E-01
PCB (mg/L)	4.00E-03	4.00E-03	4.00E-03	4.00E-03	4.00E-03	1.31E-05

Table 4-2. Best Estimate Initial Leachate Concentrations (C_0) for Waste C Alternatives.

Alternative	7	8	9	10	11	12	15	16
Treatment	No Treatment	No Treatment	Fixation	Vitrification	No Treatment	Fixation	Fixation	Fixation
Barrier	No Barrier	Hanford	RCRA	No Barrier	Hanford	Hanford	Hanford	No Barrier
Liner	No Liner	No Liner	MTR	No Liner	Single Liner	Single Liner	No Liner	No Liner
Volume Reduction	1.00	0.20	1.0	0.20	1.00	0.20	1.00	1.00
Constituent	Initial Leachate Concentrations							
U-238 (pCi/L)	1.68E+03	1.68E+03	1.68E+03	1.60E+03	1.68E+03	1.68E+03	1.68E+03	1.68E+03
Sr-90 (pCi/L)	1.25E+04	6.23E+04	1.25E+04	2.05E+02	1.25E+04	6.23E+04	1.25E+04	1.25E+04
Pu-239 (pCi/L)	2.00E+02	1.00E+03	2.00E+02	3.29E+00	2.00E+02	1.00E+03	2.00E+02	2.00E+02
CR(VI) (mg/L)	5.00E+00	5.00E+00	5.00E+00	5.00E+00	5.00E+00	5.00E+00	5.00E+00	5.00E+00
TCE (mg/L)	1.89E+01	9.47E+01	1.89E+01	3.11E-01	1.89E+01	9.47E+01	1.89E+01	1.89E+01
PCB (mg/L)	4.00E-03	2.00E-02	4.00E-03	6.56E-05	4.00E-03	2.00E-02	4.00E-03	4.00E-03

Table 4-3. Best Estimate Initial Leachate Concentrations (C_0) for Waste A Alternatives.

Alternative	13	14
Treatment	No Treatment	Vitrification
Barrier	No Barrier	RCRA
Liner	No Liner	Vault
Volume Reduction	1.00	0.20
Constituent	Initial Leachate Concentrations	
U-238 (pCi/L)	1.68E+03	1.68E+03
Sr-90 (pCi/L)	1.25E+04	6.40E+02
Pu-239 (pCi/L)	2.00E+02	1.03E+01
Cr(VI) (mg/L)	0.00E+00	0.00E+00
TCE (mg/L)	0.00E+00	0.00E+00
PCB (mg/L)	0.00E+00	0.00E+00

Table 4-4. Best Estimate (Deterministic) Results for E-waste Alternatives at C_1 (Beneath the Trench).

Alternative (Scenario)	1	2	3	4	5	6
Treatment	No Treatment	No Treatment	No Treatment	No Treatment	Fixation	Vitrification
Barrier	No Barrier	Hanford	RCRA	RCRA	RCRA	No Barrier
Liner	No Liner	No Liner	MTR	Single Liner	MTR	No Liner
Volume Reduction	1.00	1.00	1.00	1.00	1.00	1.00
Contaminant Specific ICR						
Uranium Travel Time (yr)	0	0	5,282	5,282	5,282	0
Uranium Risk	1.04E-03	1.04E-03	1.04E-03	1.04E-03	1.04E-03	1.04E-03
Strontium Travel Time (yr)	0	0	167	167	167	0
Strontium Risk	9.85E+01	9.85E+01	1.72E+00	1.72E+00	1.72E+00	3.23E-01
Plutonium Travel Time (yr)	0	0	256,564	256,564	256,564	0
Plutonium Risk	1.02E-01	1.02E-01	6.47E-05	6.47E-05	6.47E-05	3.35E-04
TCE Travel Time (yr)	0	0	2,718	2,718	2,718	0
TCE Risk	8.63E-02	8.63E-02	8.26E-61	8.26E-61	8.26E-61	2.84E-04
PCB Travel Time (yr)	0	0	128,359	128,359	128,359	0
PCB Risk	8.87E-04	8.87E-04	0.00	0.00	0.00	2.91E-06
Total ICR						
Risk at 100 years	9.86E+01	9.86E+01	0.00	0.00	0.00	3.25E-01
Risk at 1000 years	9.86E+01	9.86E+01	1.72E+00	1.72E+00	1.72E+00	3.25E-01
Risk at 10000 years	9.86E+01	9.86E+01	1.72E+00	1.72E+00	1.72E+00	3.25E-01
Maximum Risk	9.86E+01	9.86E+01	1.72E+00	1.72E+00	1.72E+00	3.25E-01
Hazard Quotient						
Chromium Travel Time (yr)	0	0	159	159	159	0
Chromium Hazard Quotient	6.10E+01	6.10E+01	6.10E+01	6.10E+01	6.10E+01	6.10E+01

Table 4-5. Best Estimate (Deterministic) Results for E-waste Alternatives at C_2 (at Water Table).

Alternative (Scenario)	1	2	3	4	5	6
Treatment	No Treatment	No Treatment	No Treatment	No Treatment	Fixation	Vitrification
Barrier	No Barrier	Hanford	RCRA	RCRA	RCRA	No Barrier
Liner	No Liner	No Liner	MTR	Single Liner	MTR	No Liner
Volume Reduction	1.00	1.00	1.00	1.00	1.00	1.00
Contaminant Specific ICR						
Uranium Travel Time (yr)	19,740	182,292	131,874	131,874	131,874	19,740
Uranium Risk	8.13E-04	1.84E-04	3.13E-04	3.13E-04	3.13E-04	8.13E-04
Strontium Travel Time (yr)	154,140	1,423,431	988,660	988,660	988,660	154,140
Strontium Risk	0.00	0.00	0.00	0.00	0.00	0.00
Plutonium Travel Time (yr)	960,540	8,870,262	6,416,465	6,416,465	6,416,465	960,540
Plutonium Risk	8.51E-14	0.00	0.00	0.00	0.00	2.80E-16
TCE Travel Time (yr)	10,140	93,639	67,745	67,745	67,745	10,140
TCE Risk	0.00	0.00	0.00	0.00	0.00	0.00
PCB Travel Time (yr)	480,540	4,437,624	3,210,041	3,210,041	3,210,041	480,540
PCB Risk	0.00	0.00	0.00	0.00	0.00	0.00
Total ICR						
Risk at 100 years	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 1000 years	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 10000 years	0.00	0.00	0.00	0.00	0.00	0.00
Maximum Risk	8.13E-04	1.84E-04	3.13E-04	3.13E-04	3.13E-04	8.13E-04
Hazard Quotient						
Chromium Travel Time (yr)	559	5,164	3,745	3,745	3,745	559
Chromium Hazard Quotient	4.77E+01	1.08E+01	1.84E+01	1.84E+01	1.84E+01	4.77E+01

Table 4-6. Best Estimate (Deterministic) Results for E-waste Alternatives at C_3 (at the ERDF Boundary).

Alternative (Scenario)	1	2	3	4	5	6
Treatment	No Treatment	No Treatment	No Treatment	No Treatment	Fixation	Vitrification
Barrier	No Barrier	Hanford	RCRA	RCRA	RCRA	No Barrier
Liner	No Liner	No Liner	MTR	Single Liner	MTR	No Liner
Volume Reduction	1.00	1.00	1.00	1.00	1.00	1.00
Contaminant Specific ICR						
Uranium Travel Time (yr)	19,745	182,297	131,879	131,879	131,879	19,745
Uranium Risk	5.90E-05	7.68E-06	1.37E-05	1.37E-05	1.37E-05	5.90E-05
Strontium Travel Time (yr)	154,174	1,423,465	988,695	988,695	988,695	154,174
Strontium Risk	0.00	0.00	0.00	0.00	0.00	0.00
Plutonium Travel Time (yr)	960,750	8,870,472	6,416,675	6,416,675	6,416,675	960,750
Plutonium Risk	6.14E-15	0.00	0.00	0.00	0.00	2.02E-17
TCE Travel Time (yr)	10,143	93,643	67,748	67,748	67,748	10,143
TCE Risk	0.00	0.00	0.00	0.00	0.00	0.00
PCB Travel Time (yr)	480,645	4,437,730	3,210,146	3,210,146	3,210,146	480,645
PCB Risk	0.00	0.00	0.00	0.00	0.00	0.00
Total ICR						
Risk at 100 years	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 1000 years	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 10000 years	0.00	0.00	0.00	0.00	0.00	0.00
Maximum Risk	5.90E-05	7.68E-06	1.37E-05	1.37E-05	1.37E-05	5.90E-05
Hazard Quotient						
Chromium Travel Time (yr)	560	5,165	3,746	3,746	3,746	560
Chromium Hazard Quotient	3.47E+00	4.51E-01	8.03E-01	8.03E-01	8.03E-01	3.47E+00

Table 4-7. Best Estimate (Deterministic) Results for C-waste Alternatives at C_1 (Beneath the Trench).

Alternative (Scenario)	7	8	9	10	11	12	15	16
Treatment	No Treatment	No Treatment	Fixation	Vitrification	No Treatment	Fixation	Fixation	Fixation
Barrier	No Barrier	Hanford	RCRA	No Barrier	Hanford	Hanford	Hanford	No Barrier
Liner	No Liner	No Liner	MTR	No Liner	Single Liner	Single Liner	No Liner	No Liner
Volume Reduction	1.00	0.20	1.00	0.20	1.00	0.20	1.00	1.00
Contaminant Specific ICR								
Uranium Travel Time (yr)	0	0	5,282	0	5,282	5,282	0	0
Uranium Risk	1.04E-03	1.04E-03	1.04E-03	9.92E-04	1.04E-03	1.04E-03	1.04E-03	1.04E-03
Strontium Travel Time (yr)	0	0	167	0	167	167	0	0
Strontium Risk	9.85E-03	4.92E-02	1.72E-04	1.62E-04	1.72E-04	8.59E-04	9.85E-03	9.85E-03
Plutonium Travel Time (yr)	0	0	256,564	0	256,564	256,564	0	0
Plutonium Risk	1.02E-03	5.10E-03	6.47E-07	1.68E-05	6.47E-07	3.24E-06	1.02E-03	1.02E-03
TCE Travel Time (yr)	0	0	2,718	0	2,718	2,718	0	0
TCE Risk	8.63E-03	4.32E-02	8.26E-62	1.42E-04	8.26E-62	4.13E-61	8.63E-03	8.63E-03
PCB Travel Time (yr)	0	0	128,359	0	128,359	128,359	0	0
PCB Risk	8.87E-04	4.44E-03	0.00	1.46E-05	0.00	0.00	8.87E-04	8.87E-04
Total ICR								
Risk at 100 years	2.14E-02	1.03E-01	0.00	1.33E-03	0.00	0.00	2.14E-02	2.14E-02
Risk at 1000 years	2.14E-02	1.03E-01	1.72E-04	1.33E-03	1.72E-04	8.59E-04	2.14E-02	2.14E-02
Risk at 10000 years	2.14E-02	1.03E-01	1.21E-03	1.33E-03	1.21E-03	1.90E-03	2.14E-02	2.14E-02
Maximum Risk	2.14E-02	1.03E-01	1.21E-03	1.33E-03	1.21E-03	1.90E-03	2.14E-02	2.14E-02
Hazard Quotient								
Chromium Travel Time (yr)	0	0	159	0	159	159	0	0
Chromium Hazard Quotient	6.10E+01	6.10E+01	6.10E+01	6.10E+01	6.10E+01	6.10E+01	6.10E+01	6.10E+01

Table 4-8. Best Estimate (Deterministic) Results for C-waste Alternatives at C_2 (at the Water Table).

Alternative (Scenario)	7	8	9	10	11	12	15	16
Treatment	No Treatment	No Treatment	Fixation	Vitrification	No Treatment	Fixation	Fixation	Fixation
Barrier	No Barrier	Hanford	RCRA	No Barrier	Hanford	Hanford	Hanford	No Barrier
Liner	No Liner	No Liner	MTR	No Liner	Single Liner	Single Liner	No Liner	No Liner
Volume Reduction	1.00	0.20	1.00	0.20	1.00	0.20	1.00	1.00
Contaminant Specific ICR								
Uranium Travel Time (yr)	19,740	182,292	131,874	19,740	242,408	242,408	182,292	19,740
Uranium Risk	8.13E-04	1.84E-04	3.13E-04	7.76E-04	1.84E-04	1.84E-04	1.84E-04	8.13E-04
Strontium Travel Time (yr)	154,140	1,423,431	988,660	154,140	1,851,765	1,851,765	1,423,431	154,140
Strontium Risk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Plutonium Travel Time (yr)	960,540	8,870,262	6,416,465	960,540	11,794,993	11,794,993	8,870,262	960,540
Plutonium Risk	8.51E-16	0.00	0.00	1.40E-17	0.00	0.00	0.00	8.51E-16
TCE Travel Time (yr)	10,140	93,639	67,745	10,140	124,524	124,524	93,639	10,140
TCE Risk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PCB Travel Time (yr)	480,540	4,437,624	3,210,041	480,540	5,900,817	5,900,817	4,437,624	480,540
PCB Risk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total ICR								
Risk at 100 years	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 1000 years	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 10000 years	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Maximum Risk	8.13E-04	1.84E-04	3.13E-04	7.76E-04	1.84E-04	1.84E-04	1.84E-04	8.13E-04
Hazard Quotient								
Chromium Travel Time (yr)	559	5,164	3,745	559	6,876	6,876	5,164	559
Chromium Hazard Quotient	4.77E+01	1.08E+01	1.84E+01	4.77E+01	1.08E+01	1.08E+01	1.08E+01	4.77E+01

Table 4-9. Best Estimate (Deterministic) Results for C-waste Alternatives at C_3 (at the ERDF Boundary).

Alternative (Scenario)	7	8	9	10	11	12	15	16
Treatment	No Treatment	No Treatment	Fixation	Vitrification	No Treatment	Fixation	Fixation	Fixation
Barrier	No Barrier	Hanford	RCRA	No Barrier	Hanford	Hanford	Hanford	No Barrier
Liner	No Liner	No Liner	MTR	No Liner	Single Liner	Single Liner	No Liner	No Liner
Volume Reduction	1.00	0.20	1.00	0.20	1.00	0.20	1.00	1.00
Contaminant Specific ICR								
Uranium Travel Time (yr)	19,745	182,297	131,879	19,745	242,413	242,413	182,297	19,745
Uranium Risk	5.90E-05	7.68E-06	1.37E-05	5.64E-05	7.68E-06	7.68E-06	7.68E-06	5.90E-05
Strontium Travel Time (yr)	154,174	1,423,465	988,695	154,174	1,851,799	1,851,799	1,423,465	154,174
Strontium Risk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Plutonium Travel Time (yr)	960,750	8,870,472	6,416,675	960,750	11,795,202	11,795,202	8,870,472	960,750
Plutonium Risk	6.14E-17	0.00	0.00	1.01E-18	0.00	0.00	0.00	6.14E-17
TCE Travel Time (yr)	10,143	93,643	67,748	10,143	124,527	124,527	93,643	10,143
TCE Risk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
PCB Travel Time (yr)	480,645	4,437,730	3,210,146	480,645	5,900,922	5,900,922	4,437,730	480,645
PCB Risk	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Total ICR								
Risk at 100 years	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 1000 years	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Risk at 10000 years	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
Maximum Risk	5.90E-05	7.68E-06	1.37E-05	5.64E-05	7.68E-06	7.68E-06	7.68E-06	5.90E-05
Hazard Quotient								
Chromium Travel Time (yr)	560	5,165	3,746	560	6,877	6,877	5,165	560
Chromium Hazard Quotient	3.47E+00	4.51E-01	8.03E-01	3.47E+00	4.51E-01	4.51E-01	4.51E-01	3.47E+00

Table 4-10. Best Estimate (Deterministic) Results for A-waste at C_1, C_2, and C_3.

Compliance Point	Beneath the Trench (C_1)		At the Water Table (C_2)		At the ERDF Boundary (C_3)	
Alternative (Scenario)	13	14	13	14	13	14
Treatment	No Treatment	Vitrification	No Treatment	Vitrification	No Treatment	Vitrification
Barrier	No Barrier	RCRA	No Barrier	RCRA	No Barrier	RCRA
Liner	No Liner	Vault	No Liner	Vault	No Liner	Vault
Volume Reduction	1.00	0.20	1.00	0.20	1.00	0.20
Contaminant Specific ICR						
Uranium Travel Time (yr)	0	2,004	19,740	135,450	19,745	135,455
Uranium Risk	1.04E-03	1.04E-03	8.13E-04	3.13E-04	5.90E-05	1.37E-05
Strontium Travel Time (yr)	0	16,004	154,140	1,058,018	154,174	1,058,053
Strontium Risk	9.85E-03	0.00	0.00	0.00	0.00	0.00
Plutonium Travel Time (yr)	0	100,004	960,540	6,593,426	960,750	6,593,636
Plutonium Risk	1.02E-03	2.97E-06	8.51E-16	0.00	6.14E-17	0.00
Total ICR						
Risk at 100 years	1.19E-02	0.00	0.00	0.00	0.00	0.00
Risk at 1000 years	1.19E-02	0.00	0.00	0.00	0.00	0.00
Risk at 10000 years	1.19E-02	1.04E-03	0.00	0.00	0.00	0.00
Maximum Risk	1.19E-02	1.04E-03	8.13E-04	3.13E-04	5.90E-05	1.37E-05

Table 4-11. Alternative Scores and Relative Rank for Risk at the ERDF Boundary Performance Measures.

Alternative	Median Maximum Risk	95th Percentile for Maximum Risk	Median Travel Time of U-238 (Years)	95th Percentile for 10,000 Year Risk	Probability that Maximum Risk is less than 10^{-5} (percent)
E-type Waste					
1	2.6E-4 (6)	1.7E-3 (6)	11,000 (5t)	8.2E-4 (6)	0.5 (5t)
2	1.1E-5 (1)	7.3E-5 (1)	240,000 (4)	0 (1t)	48 (1)
3	1.4E-5 (2t)	8.7E-5 (2t)	250,000 (1t)	0 (1t)	39 (2t)
4	1.4E-5 (2t)	8.7E-5 (2t)	250,000 (1t)	0 (1t)	39 (2t)
5	1.4E-5 (2t)	8.7E-5 (2t)	250,000 (1t)	0 (1t)	39 (2t)
6	1.9E-4 (5)	7.5E-4 (5)	11,000 (5t)	6.5E-4 (5)	0.5 (5t)
C-type Waste					
7	2.2E-4 (6t)	1.0E-3 (7t)	11,000 (6t)	8.2E-4 (7t)	0.5 (7t)
8	1.1E-5 (1t)	7.3E-5 (1t)	240,000 (4t)	0 (1t)	48 (1t)
9	1.4E-5 (5)	8.7E-5 (5)	250,000 (3)	0 (1t)	39 (5)
10	2.4E-5 (8)	1.3E-4 (6)	11,000 (6t)	1.0E-4 (6)	23 (6)
11	1.1E-5 (1t)	7.3E-5 (1t)	320,000 (1t)	0 (1t)	48 (1t)
12	1.1E-5 (1t)	7.3E-5 (1t)	320,000 (1t)	0 (1t)	48 (1t)
15	1.1E-5 (1t)	7.3E-5 (1t)	240,000 (4t)	0 (1t)	48 (1t)
16	2.2E-4 (6t)	1.0E-3 (7t)	11,000 (6t)	8.2E-4 (7t)	0.5 (7t)
A-type Waste					
13	2.2E-4 (2)	1.0E-3 (2)	11,000 (2)	8.2E-4 (2)	0.5 (2)
14	1.0E-5 (1)	5.9E-5 (1)	260,000 (1)	0 (1)	52 (1)

Notes:
Rank shown in parentheses.
t - Tie rank.

Table 4-12. Alternative Scores and Relative Rank for Hazard Quotient at the ERDF Boundary Performance Measures.

Alternative	Median Hazard Quotient	95th Percentile for Hazard Quotient	Median Travel Time of Cr(VI) (Years)	95th Percentile for 10,000 Year Hazard Quotient	Probability that Hazard Quotient is less than 1 (percent)
E-type Waste					
1	7.2 (6)	26 (6)	360 (5t)	26 (6)	4 (5t)
2	3.4E-1 (1)	1.8 (1)	7,400 (4)	1.8 (1)	85 (1)
3	4.4E-1 (2t)	2.3 (2t)	8,000 (1t)	2.3 (2t)	79 (2t)
4	4.4E-1 (2t)	2.3 (2t)	8,000 (1t)	2.3 (2t)	79 (2t)
5	4.4E-1 (2t)	2.3 (2t)	8,000 (1t)	2.3 (2t)	79 (2t)
6	6.9 (5)	25 (5)	360 (5t)	25 (5)	4 (5t)
C-type Waste					
7	7.2 (7t)	26 (7t)	360 (6t)	26 (7t)	4 (7t)
8	3.4E-1 (1t)	1.8 (1t)	7,400 (4t)	1.8 (1t)	85 (1t)
9	4.4E-1 (5)	2.3 (5)	8,000 (3)	2.3 (5)	79 (5)
10	1.4 (6)	5.6 (6)	360 (6t)	5.6 (6)	38 (6)
11	3.4E-1 (1t)	1.8 (1t)	9,700 (1t)	1.8 (1t)	85 (1t)
12	3.4E-1 (1t)	1.8 (1t)	9,700 (1t)	1.8 (1t)	85 (1t)
15	3.4E-1 (1t)	1.8 (1t)	7,400 (4t)	1.8 (1t)	85 (1t)
16	7.2 (7t)	26 (7t)	360 (6t)	26 (7t)	4 (7t)

Notes:
Rank shown in parentheses.
t - Tie rank.

5.0 SUMMARY

As shown in Table 1-1, 16 different design and treatment scenarios were simulated in this study. The elements that varied between the scenarios included the waste type, the treatment approach, the liner, and the barrier. This study was intended to be a scoping study that relied upon simple analytical tools and readily available information regarding governing processes and parameters. A spreadsheet model was created that relied upon simple analytical expressions for description of contaminant fate and transport. In general, these expressions assume steady-state flow conditions and one-dimensional plug flow.

Performance was measured in terms of maximum risk and travel time at three points of compliance: 1) the bottom of the trench (C_1), 2) the water table directly beneath the ERDF (C_2), and 3) the saturated zone intercepted by a vertical plane extending down from the boundary of the ERDF (C_3). The exposure assessment assumed residential consumption of groundwater at each point of compliance. Other exposure pathways were not addressed. Risk calculations were performed using the approach described in the Hanford Site Baseline Risk Assessment Methodology (HSBRAM) (DOE-RL 1993). With the exception of Cr(VI), all the simulated contaminants are carcinogens and their health effects can be expressed in terms of an ICR. Because Cr(VI) is a chemical toxin but not a carcinogen (via ingestion), it was necessary to express Cr(VI) health effects in terms of a hazard quotient. Although it was assumed that Cr(VI) would be a significant contaminant of concern and that Cr(VI) would not be reduced to Cr(III), the presence of Cr(VI) in 100 Area and 300 Area soils has not been verified.

For the compliance point directly beneath the trench, it was necessary to assume that vadose zone pore water is collected and consumed by the exposed individual. Although this pathway is not real, it provides a performance measure in relation to the "no-release" criteria. Even the other two compliance points are unrealistic since no wells are located near the proposed facility and it is unlikely that water supply wells will be completed beneath or near the facility as long as the 200 Area remains under institutional control. Given these unrealistic exposure pathways, the risk results presented in this study should be considered performance measures against different criteria, not actual risk estimates. This point is further emphasized by the fact that not all exposure pathways were considered, only six contaminants were addressed, and actual concentrations of those six contaminants will likely differ from those assumed in the simulations.

Both deterministic and Monte Carlo simulations were performed. The results of the Monte Carlo simulations suggest the following conclusions regarding estimated risks:

- Estimated maximum risks directly beneath the trench (at C_1) are high for all the alternatives, with median ICRs ranging from 10^{-4} to greater than 1, and median HQs ranging from 10 to 100.
- Sr-90, TCE, and uranium were generally the prime cancer-causing risk drivers beneath the trench.
- Median ICRs at the ERDF boundary for all the alternatives are in the range of 10^{-5} to 10^{-3}, and are expected to occur between 10,000 and

500,000 years in the future. Uranium is the only cancer-causing contaminant that reached groundwater at significant risk levels.

- Assuming that Cr(VI) is a contaminant, median HQs at the ERDF boundary are in the range of 0.5 to 10 for all the alternatives, and are expected to occur between 500 and 10,000 years in the future.
- None of the cancer-causing constituents were predicted to reach groundwater within 10,000 years for the barrier alternatives; therefore, it is highly likely that the ICR in groundwater for the barrier alternatives will be negligible for the next 10,000 years. For the alternatives without barriers, the probability that uranium would reach groundwater within 10,000 years was approximately 50 percent; in realizations where U-238 reached groundwater the ICR exceeded 10^{-4}.

The results suggested the following conclusions regarding the relative performance of the various configuration elements:

- Because leachate concentrations were limited by solubility or sorption controls, fixation did not result in any reduction in risk compared with untreated waste.
- An infiltration barrier is probably the most critical element of the facility performance.
- Differences between the performance of the modified RCRA barrier and the Hanford barrier were virtually indistinguishable.
- Due to decay during migration through the liner, the presence of a liner reduced the ICR at the base of the trench but did not change the HQ (chromium(VI) was assumed not to decay). However, liners did not significantly change the ICRs in the groundwater.
- At C_1, vitrification reduced the ICR for E waste and reduced both the ICR and the HQ for C waste. At C_2 and C_3, vitrification only reduced the HQ for C waste alternatives.

The parameter sensitivity analyses indicated that U-238 solubility, barrier infiltration rate, saturated zone hydraulic conductivity, saturated zone hydraulic gradient, and the vadose zone mixing factor are the most important parameters for determining maximum risk. For alternatives that include vitrification, the dissolution rate of vitrified material and the U-238 K_d are also important. In general, the list of important parameters is reduced when comparing relative performance of alternatives.

Finally, general conclusions resulting from this study include the following:

- The presence of Cr(VI) as a significant contaminant in 100 Area and 300 Area soils should be determined.

- The performance criteria (including time and place of compliance and allowable risk) should be determined as soon as possible to help focus future performance assessments and conceptual design of the ERDF.
- Although the benefits of fixation and vitrification are not warranted by the increase in cost, other types of treatment that result in chemical changes affecting contaminant mobility or solubility may deserve further attention. Because the effectiveness of these types of treatment are dependent on waste characteristics, they should be evaluated in the operable unit RI/FS process.

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APPENDIX A

HELP SIMULATIONS OF ALTERNATIVE BARRIERS

A.1 Objectives

Characterization of the performance of the modified RCRA and Hanford barriers is an important part of this risk modeling effort. The infiltration rates selected for these barriers greatly affect the behavior of the model. Vadose zone travel time for infiltrating moisture is related to the infiltration rate. The vadose zone travel time is a controlling factor for concentrations of contaminants at the point water reaches the water table. A small infiltration rate results in a longer vadose zone travel time and lower concentrations of contaminants at the point they enter the water table (longer time to degrade). The HELP 2.05 model was used as a tool to assist with the evaluation of the performance of these barriers.

A.2 General Approach

The HELP model is a quasi-two-dimensional hydrologic model of water movement across, into, through, and out of landfills (Schroeder 1988). HELP was developed by the U.S. Army Engineer Waterways Experimental Station (WES) for the U.S. Environmental Protection Agency, and is widely used for the evaluation of landfill design. Although HELP is a popular code, it does have some limitations. Among these are the models inability to model capillary break flow and model asphaltic concrete layers. Both of these facts make the HELP model "conservative" in that it will over-predict actual percolation through the landfill cap.

The HELP model (version 2.05) was used to examine the characterization of the hydrologic performance of the Hanford and modified RCRA barriers at the ERDF site. To ensure the model and the computer were working correctly, an example problem with known results was provided by the author of the code. The input files provided were run, producing results duplicating the example output provided by the author. This indicated that both the hardware and software used were functioning correctly.

It must be noted that this effort simply evaluates the design performance of the barriers. Changes in the hydrologic performance of the barriers due to construction defects, intrusion or weathering are not evaluated, as this is beyond the scope of this task. Qualitatively, it is reasonable to assume that the Hanford barrier will out perform the modified RCRA barrier in the future, due to greater thickness and design redundancy. However, without a detailed analysis, it is impossible to determine to what extent this might occur, and the time frame associated with it.

A.3 Input Data Used in both the Hanford Barrier and the Modified RCRA Barrier

Values for porosity, saturated hydraulic conductivity, field capacity and wilting point for the soils (McGee Ranch silt) to be used in the top two layers of both the modified RCRA barrier and the Hanford barrier were obtained from lab data gathered and calculations performed by Pacific Northwest Laboratory (Nichols 1991). The values were modified for the top layer to reflect the addition of pea gravel, and modified for the second layer to reflect the effects of soil compaction.

The saturated hydraulic conductivity value for the top layer was not changed from the unmodified McGee Ranch silt value. It is thought that the addition of the pea gravel should not affect the conductivity (due to the fact that not enough gravel is added to result in grain to grain contact). The values for porosity, field capacity and wilting point were all reduced by 7.2 percent to account for the addition of the pea gravel.

The value for saturated hydraulic conductivity of the compacted McGee Ranch silt used was 1.6×10^{-6} cm/sec (personal communication with W.A. Skelly of Westinghouse). Porosity and field capacity of the compacted silt were reduced 25 percent below that in the uncompacted silt, consistent with the method used by HELP to account for compaction. The wilting point was reduced by 26.5 percent. There were no guiding principles available to adjust the value of wilting point for the effects of compaction. The wilting point was adjusted to maintain approximately the same absolute difference and percentage with field capacity, as the relationship exists between the two in the top admix layer.

The required solar radiation and temperature data files for a 10-year period were synthetically created from actual mean and standard deviation data from the Hanford Meteorological Station (HMS) near the 200 Area. Actual precipitation data from the HMS (January 79 through December 88) were used to run the model. Both barriers were modeled with a "Poor Grass" vegetative cover (Leaf Area Index = 1.6) and a growing season beginning on Julian day 113 and ending on Julian day 288 (Approximately, mid-April through mid October). All runs were conducted with an evaporative zone of 91 cm (36 in.), unless otherwise stated.

A.4 RCRA Barrier Simulations

The modified RCRA barrier was modeled as designed, with six layers. The top four layers were modeled as vertical percolation layers, layer five was modeled as a lateral drainage layer and layer six was modeled as a barrier soil layer. Although layer six is a surface treated asphaltic concrete layer, the HELP model does not recognize asphaltic concrete as a layer material, so the layer had to be modeled as a barrier soil, with very low hydraulic conductivity, porosity, field capacity and wilting point. The treated asphalt will probably resist infiltration more effectively than a barrier soil. This fact makes it likely that the model overestimates leakage through layer six.

A.4.1 Comparison with Hanford Modeling

The model simulations were initially compared with results generated by WHC in an unpublished report, using similar input data. The only data that could not be verified to be exactly the same were the temperature, solar radiation and precipitation files. The results of the comparison showed very close agreement (within 5 percent) for average annual runoff, evapotranspiration, and percolation through layer 6. Close agreement was obtained for average annual change in soil water storage (Westinghouse's 0.0 cm vs 7.6×10^{-3} cm). It is thought that the result differences may be accounted for by differences in the precipitation, temperature and solar radiation files. The precipitation file used by Westinghouse reported an average annual precipitation of 17.8 cm (7.00 in.) for the period from January 1, 1979 through December 31, 1988. The actual average for the period is 18.0 cm (7.08 in.) (Westinghouse Engineering Data Transmittal SD-EN-CSWD-028) and that was the average value of the input data used in this study. Although the stochastic generator used by the model created a data set with the same mean and standard deviation as the data used by Westinghouse, each data set is unique, and likely had differences between them.

A.4.2 Simulations Using Best Estimates for Input Data

The basis for WHC soil parameters for the top two layers could not be ascertained, and were consequently disregarded in further modeling. Instead, the top two layers were modeled using soil parameters obtained from Nichols (1991). Layer 3 was specified as a compacted HELP soil type 3 (filter sand layer). Layer 4 was specified as a compacted HELP soil type 1 (filter gravel layer). Layer 5 was specified as an uncompacted HELP soil type 1 with increased saturated hydraulic conductivity (drainage gravel layer). Layer 6 was specified as a barrier soil with very low saturated hydraulic conductivity, porosity, field capacity and wilting point. Soil parameters for layers 3 through 6 were specified the same as the layers in the WHC modified RCRA barrier model.

The initial soil water content of each layer was determined by assigning the model arbitrary initial water contents for each layer, running the model over the 10-year period, and using the final water content as the initial soil water content for the next run. This was performed until a steady-state initial soil water content was established, and there was little or no change in water storage over a 10-year period. After satisfactory initial water contents had been established the model was run for a 10-year period. The model showed zero average annual percolation though layer 6 over the 10-year period 1979 through 1988. Most of the precipitation (99.99 percent) was evaporated and transpired out of the system. The remaining water (0.01 percent) ran off the barrier.

A.4.3 Simulation of Extreme Precipitation Events

To test the effects of increased precipitation on the barrier, two 13 cm (5 in.) artificial precipitation events were added in the spring of 1983 to the input precipitation file. For the first run, the water contents of each soil layer was initialized with the storm events in the precipitation file. This was done to simulate a relatively minor increase of average annual precipitation 2.5 cm (1 in.) extra per year. The results of this simulation showed that percolation through the bottom of the barrier would occur with the increase in average annual precipitation (0.16 mm/yr vs. 0.00 mm/yr). The effects that these two isolated extreme events would have on the barriers with water contents initialized using the actual precipitation file was then tested. Under these conditions, the average annual percolation through layer 6 remained 0.00 mm/yr. These results indicate that the barrier should perform very effectively with rare extreme precipitation events, but performance probably will decrease if long term precipitation averages increase.

A.5 Hanford Barrier Simulations

The Hanford barrier was modeled as designed, with seven layers. The top five layers were modeled as vertical percolation layers, layer six was modeled as a lateral drainage layer, and layer seven was modeled as a barrier soil. Layer seven is a surface treated asphaltic concrete layer, but was treated as a barrier soil, due to the model's inability to model asphaltic concrete. Because the treated asphalt will likely resist infiltration more effectively than barrier soil, it is likely that the model will overestimate leakage through layer seven. Another reason it is thought that HELP will overestimate the amount of infiltration is the fact that the model does not model capillary breaks, such as the one that occurs between layer 5 and layer 6.

A.5.1 Comparison with Hanford Modeling

As with the RCRA barrier, an attempt was made to simulate the results of the unpublished WHC report. This effort produced results very similar to those produced by Westinghouse. The run produced average annual values for runoff, evapotranspirations, lateral drainage from layer 6, and percolation through layer 7 of 2.5×10^{-3} cm (1×10^{-3} in.), 18 cm (7 in.), 7.6×10^{-4} cm (3×10^{-4} in.), and 4.6×10^{-2} cm (1.8×10^{-2} in.), respectively compared to 2.5×10^{-3} cm (1×10^{-3} in.), 18 cm (7 in.), 7.6×10^{-4} cm (3×10^{-4} in.) and 4.6×10^{-2} cm (1.8×10^{-2} in.), respectively for Westinghouse results. The simulation run showed an average annual -3.6×10^{-2} cm (-1.4×10^{-2} in.) change in water storage, compared to 0.0 cm (0.0 in.) obtained by Westinghouse.

A.5.2 Simulations Using Best Estimates for Input Data

Layers 1 through 4 were given the same saturated hydraulic conductivity, porosity, field capacity and wilting point as the modified RCRA barrier layers 1 through 4. Layer 5 (biointrusion resistant layer) was specified as an uncompacted HELP soil type 1 with an increased saturated hydraulic conductivity value. Layers 6 and 7 were specified with the same parameters (other than thickness) as modified RCRA layers 5 and 6, respectively. Water content values were initialized by the same procedure used with the modified RCRA barrier. Surprisingly, HELP gives an average annual percolation rate through layer 7 of 1.43 mm/yr, which is quite a bit higher that the value given for the modified RCRA barrier (no predicted average annual percolation). It was reasoned that the difference of performance of the barriers with these parameters was due to the fact that the evaporative zone did not extend past the more permeable upper layer of the Hanford barrier. Additional runs for the Hanford barrier were subsequently undertaken. The next run was performed with a 51 cm (20 in.) top layer and a 152 cm (60 in.) second-layer, with all other parameters remaining constant. These layer thicknesses are reasonable to assume in that the lower part of layer 1 will likely become compacted as the top part of layer 1 is constructed. The water contents were again initialized. The results of the run showed virtually no average annual percolation through layer 7 (0.01 mm/yr). Another run was done, this time with a 112 cm (44 in.) specified evaporative zone, and all other parameters remaining equal. The water contents were initialized. Again, the results of the run showed virtually no average annual percolation through layer 7 (only 0.01 mm/yr).

A.6 Conclusions

According to the results of the modeling of the modified RCRA and Hanford barriers with HELP model version 2.05, it appears that a major factor to consider when evaluating the performance of the barriers is the depth of the evaporative zone, and the thickness of the uncompacted layer (layer 1). The modified RCRA barrier is predicted to perform perfectly for a specified evaporative zone of 91 cm (36 in.) whereas the Hanford barrier is predicted to allow 1.43 mm/yr of percolation through the barrier with that same evaporative zone depth. When the top layer of the Hanford barrier is specified as 51 cm (20 in.) and when the evaporative zone is extended to 112 cm (44 in.) on the standard Hanford barrier, the Hanford barrier performs equally well, with a negligible amount percolation through the barrier.

The results suggest that to maximize performance of the Hanford barrier, the top layer of the barrier must not extend past the evaporative zone of the vegetation. By keeping the top layer within the evaporative zone, where the saturated hydraulic conductivity is higher than the second layer, soil moisture is retained until it is used by the vegetation. If the top layer is thicker than the evaporative zone, moisture from precipitation will migrate to the lower level of the layer, beyond the vegetation's ability to access the moisture. The moisture will then percolate through the rest of the barrier.

Although the modeling of the Hanford barrier with a 91 cm (36 in.) evaporative zone and 102 cm (40 in.) uncompacted McGee Ranch admix top layer predicts 0.14 cm/yr percolation, it is reasonable to assume that the actual percolation rate will be less, based on lysimeter data from Hanford for McGee Ranch soils (Gee et al, 1992). Also, the likelihood that the lower level of layer 1 will become compacted during construction suggests that the performance of the Hanford barrier will be better than indicated by these HELP simulations.

A.7 References

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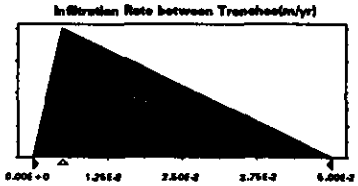
APPENDIX B

PROBABILITY DISTRIBUTIONS FOR INPUT PARAMETERS

Assumption: Infiltration Rate between Trenches(m/yr)

Triangular distribution with parameters:

Minimum	0.00E+00
Likeliest	5.00E-03
Maximum	5.00E-02



Selected range is from 0.00E+0 to 5.00E-2

Mean value in simulation was 1.59E-2

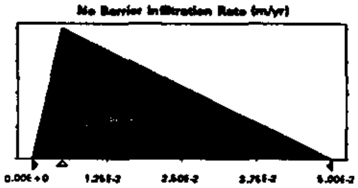
Correlated with:

No Barrier Infiltration Rate (m/yr)	0.95
RCRA Barrier Infiltration Rate (m/yr)	0.85
Hanford Barrier Infiltration Rate (m/yr)	0.85

Assumption: No Barrier Infiltration Rate (m/yr)

Triangular distribution with parameters:

Minimum	0.00E+00
Likeliest	5.00E-03
Maximum	5.00E-02



Selected range is from 0.00E+0 to 5.00E-2

Mean value in simulation was 1.81E-2

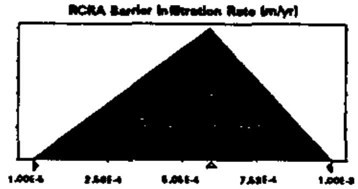
Correlated with:

Infiltration Rate between Trenches(m/yr)	0.95

Assumption: Modified RCRA Barrier Infiltration Rate (m/yr)

Triangular distribution with parameters:

Minimum	1.00E-05
Likeliest	6.00E-04
Maximum	1.00E-03



Selected range is from 1.00E-5 to 1.00E-3

Mean value in simulation was 5.61E-4

Correlated with:

Infiltration Rate between Trenches(m/yr)	0.85

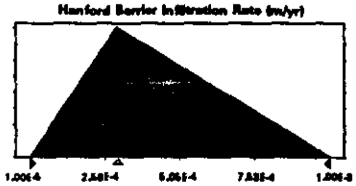
9413280.0286

9413280.0287

Assumption: Hanford Barrier Infiltration Rate (m/yr)

Triangular distribution with parameters:

Minimum	1.00E-05
Likeliest	3.00E-04
Maximum	1.00E-03



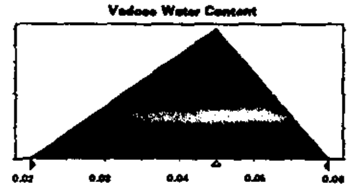
Selected range is from 1.00E-5 to 1.00E-3
Mean value in simulation was 4.22E-4
Correlated with:

Infiltration Rate between Trenches(m/yr)	0.85

Assumption: Vadose Water Content

Triangular distribution with parameters:

Minimum	0.02
Likeliest	0.04
Maximum	0.06

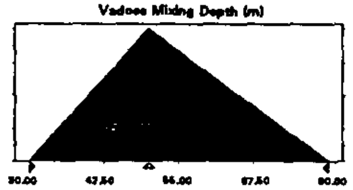


Selected range is from 0.02 to 0.06
Mean value in simulation was 0.04

Assumption: Vadose Mixing Depth (m)

Triangular distribution with parameters:

Minimum	30.00
Likeliest	50.00
Maximum	80.00

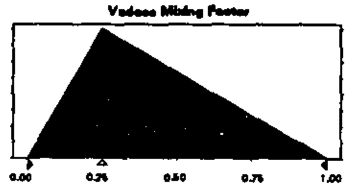


Selected range is from 30.00 to 80.00
Mean value in simulation was 52.87

Assumption: Vadose Mixing Factor

Triangular distribution with parameters:

Minimum	0.00
Likeliest	0.25
Maximum	1.00

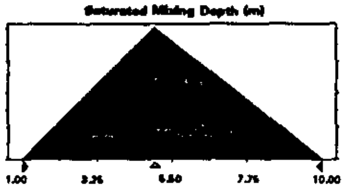


Selected range is from 0.00 to 1.00
Mean value in simulation was 0.46

Assumption: Saturated Mixing Depth (m)

Triangular distribution with parameters:

Minimum	1.00
Likeliest	5.00
Maximum	10.00



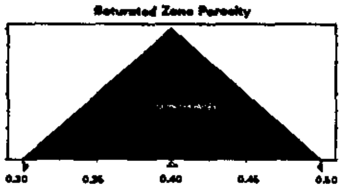
Selected range is from 1.00 to 10.00

Mean value in simulation was 5.01

Assumption: Saturated Zone Porosity

Triangular distribution with parameters:

Minimum	0.30
Likeliest	0.40
Maximum	0.50



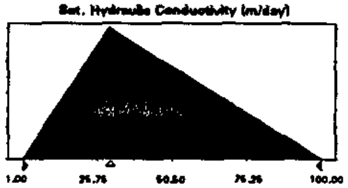
Selected range is from 0.30 to 0.50

Mean value in simulation was 0.41

Assumption: Saturated Hydraulic Conductivity (m/day)

Triangular distribution with parameters:

Minimum	1.00
Likeliest	30.00
Maximum	100.00



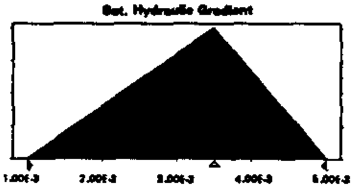
Selected range is from 1.00 to 100.00

Mean value in simulation was 47.06

Assumption: Saturated Hydraulic Gradient

Triangular distribution with parameters:

Minimum	1.00E-03
Likeliest	3.50E-03
Maximum	5.00E-03



Selected range is from 1.00E-3 to 5.00E-3

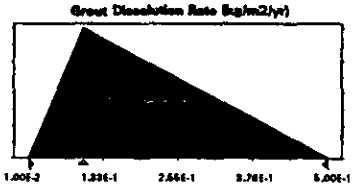
Mean value in simulation was 3.20E-3

9413280.0289

Assumption: Grout Dissolution Rate (kg/m2/yr)

Triangular distribution with parameters:

Minimum	1.00E-02
Likeliest	1.00E-01
Maximum	5.00E-01



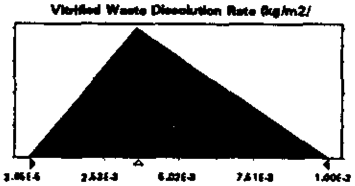
Selected range is from 1.00E-2 to 5.00E-1

Mean value in simulation was 1.82E-1

Assumption: Vitrified Waste Dissolution Rate (kg/m2/yr)

Triangular distribution with parameters:

Minimum	3.65E-05
Likeliest	3.65E-03
Maximum	1.00E-02



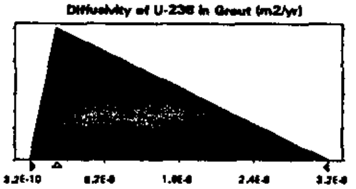
Selected range is from 3.65E-5 to 1.00E-2

Mean value in simulation was 4.96E-3

Assumption: Diffusivity of U-238 in Grout (m2/yr)

Triangular distribution with parameters:

Minimum	3.2E-10
Likeliest	3.2E-09
Maximum	3.2E-08



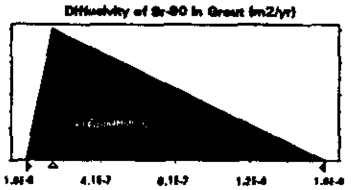
Selected range is from 3.2E-10 to 3.2E-8

Mean value in simulation was 1.2E-8

Assumption: Diffusivity of Sr-90 in Grout (m2/yr)

Triangular distribution with parameters:

Minimum	1.6E-08
Likeliest	1.6E-07
Maximum	1.6E-06



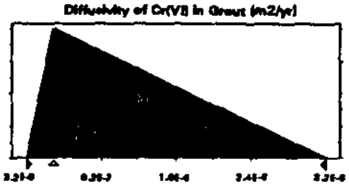
Selected range is from 1.6E-8 to 1.6E-6

Mean value in simulation was 6.2E-7

Assumption: Diffusivity of Cr(VI) in Grout (m2/yr)

Triangular distribution with parameters:

Minimum	3.2E-08
Likeliest	3.2E-07
Maximum	3.2E-06

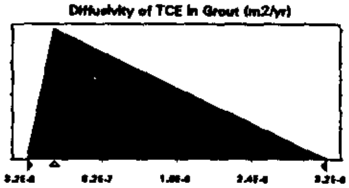


Selected range is from 3.2E-8 to 3.2E-6
Mean value in simulation was 1.3E-6

Assumption: Diffusivity of TCE in Grout (m2/yr)

Triangular distribution with parameters:

Minimum	3.2E-08
Likeliest	3.2E-07
Maximum	3.2E-06

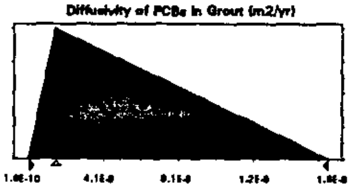


Selected range is from 3.2E-8 to 3.2E-6
Mean value in simulation was 1.2E-6

Assumption: Diffusivity of PCBs in Grout (m2/yr)

Triangular distribution with parameters:

Minimum	1.6E-10
Likeliest	1.6E-09
Maximum	1.6E-08

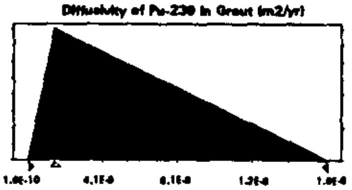


Selected range is from 1.6E-10 to 1.6E-8
Mean value in simulation was 5.9E-9

Assumption: Diffusivity of Pu-239 in Grout (m2/yr)

Triangular distribution with parameters:

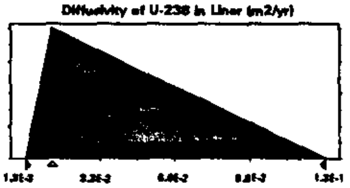
Minimum	1.6E-10
Likeliest	1.6E-09
Maximum	1.6E-08



Selected range is from 1.6E-10 to 1.6E-8
Mean value in simulation was 5.7E-9

Assumption: Diffusivity of U-238 in Liner (m2/yr)

Triangular distribution with parameters:	
Minimum	1.3E-03
Likeliest	1.3E-02
Maximum	1.3E-01

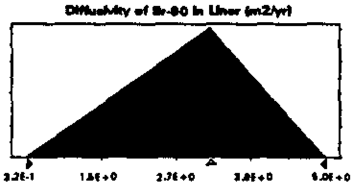


Selected range is from 1.3E-3 to 1.3E-1

Mean value in simulation was 4.6E-2

Assumption: Diffusivity of Sr-90 in Liner (m2/yr)

Triangular distribution with parameters:	
Minimum	3.2E-01
Likeliest	3.2E+00
Maximum	5.0E+00

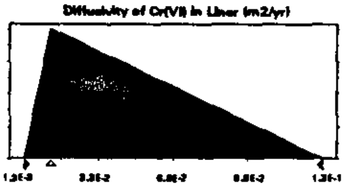


Selected range is from 3.2E-1 to 5.0E+0

Mean value in simulation was 2.9E+0

Assumption: Diffusivity of Cr(VI) in Liner (m2/yr)

Triangular distribution with parameters:	
Minimum	1.3E-03
Likeliest	1.3E-02
Maximum	1.3E-01

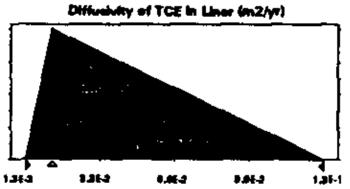


Selected range is from 1.3E-3 to 1.3E-1

Mean value in simulation was 4.9E-2

Assumption: Diffusivity of TCE in Liner (m2/yr)

Triangular distribution with parameters:	
Minimum	1.3E-03
Likeliest	1.3E-02
Maximum	1.3E-01



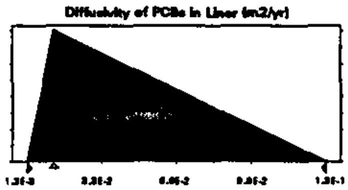
Selected range is from 1.3E-3 to 1.3E-1

Mean value in simulation was 5.0E-2

Assumption: Diffusivity of PCBs in Liner (m2/yr)

Triangular distribution with parameters:

Minimum	1.3E-03
Likeliest	1.3E-02
Maximum	1.3E-01

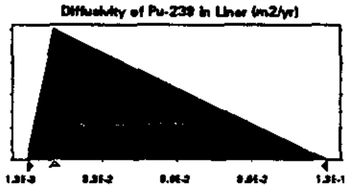


Selected range is from 1.3E-3 to 1.3E-1
Mean value in simulation was 4.3E-2

Assumption: Diffusivity of Pu-239 in Liner (m2/yr)

Triangular distribution with parameters:

Minimum	1.3E-03
Likeliest	1.3E-02
Maximum	1.3E-01

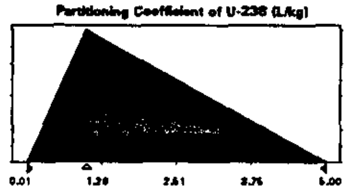


Selected range is from 1.3E-3 to 1.3E-1
Mean value in simulation was 4.8E-2

Assumption: Partitioning Coefficient of U-238 (L/kg)

Triangular distribution with parameters:

Minimum	0.01
Likeliest	1.00
Maximum	5.00

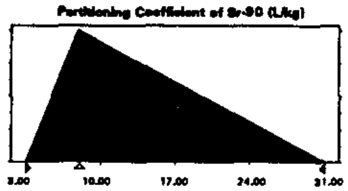


Selected range is from 0.01 to 5.00
Mean value in simulation was 1.77

Assumption: Partitioning Coefficient of Sr-90 (L/kg)

Triangular distribution with parameters:

Minimum	3.00
Likeliest	8.00
Maximum	31.00



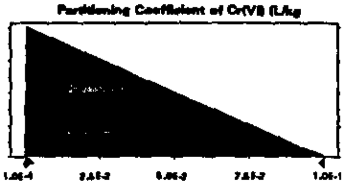
Selected range is from 3.00 to 31.00
Mean value in simulation was 13.33

9413280.0292

Assumption: Partitioning Coefficient of Cr(VI) (L/kg)

Triangular distribution with parameters:

Minimum	1.0E-04
Likeliest	1.0E-03
Maximum	1.0E-01



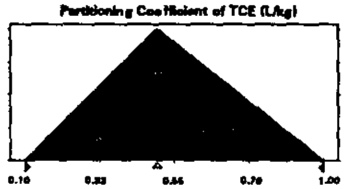
Selected range is from 1.0E-4 to 1.0E-1

Mean value in simulation was 3.5E-2

Assumption: Partitioning Coefficient of TCE (L/kg)

Triangular distribution with parameters:

Minimum	0.10
Likeliest	0.50
Maximum	1.00



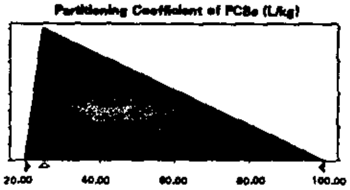
Selected range is from 0.10 to 1.00

Mean value in simulation was 0.56

Assumption: Partitioning Coefficient of PCBs (L/kg)

Triangular distribution with parameters:

Minimum	20.00
Likeliest	25.00
Maximum	100.00



Selected range is from 20.00 to 100.00

Mean value in simulation was 47.71

Assumption: Partitioning Coefficient of Pu-239 (L/kg)

Triangular distribution with parameters:

Minimum	21.00
Likeliest	50.00
Maximum	100.00

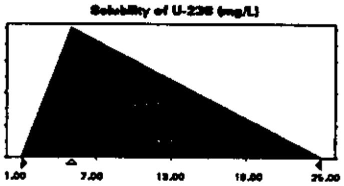
Selected range is from 21.00 to 100.00

Mean value in simulation was 60.74

Assumption: Solubility of U-238 (mg/L)

Triangular distribution with parameters:

Minimum	1.00
Likeliest	5.00
Maximum	25.00

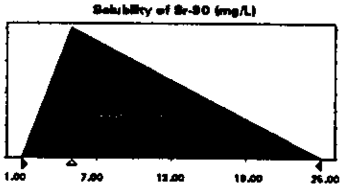


Selected range is from 1.00 to 25.00
Mean value in simulation was 11.96

Assumption: Solubility of Sr-90 (mg/L)

Triangular distribution with parameters:

Minimum	1.00
Likeliest	5.00
Maximum	25.00

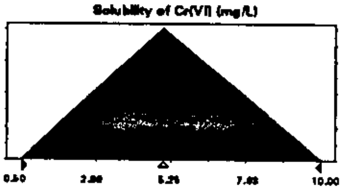


Selected range is from 1.00 to 25.00
Mean value in simulation was 9.42

Assumption: Solubility of Cr(VI) (mg/L)

Triangular distribution with parameters:

Minimum	0.50
Likeliest	5.00
Maximum	10.00

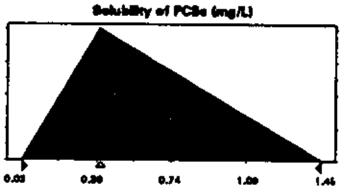


Selected range is from 0.50 to 10.00
Mean value in simulation was 5.18

Assumption: Solubility of PCBs (mg/L)

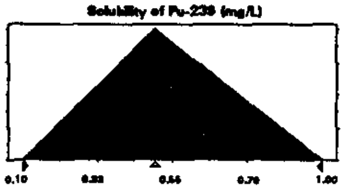
Triangular distribution with parameters:

Minimum	0.03
Likeliest	0.40
Maximum	1.45



Selected range is from 0.03 to 1.45
Mean value in simulation was 0.66

Assumption: Solubility of Pu-239 (mg/L)



Triangular distribution with parameters:	
Minimum	0.10
Likeliest	0.50
Maximum	1.00

Selected range is from 0.10 to 1.00
Mean value in simulation was 0.60

APPENDIX C

ERDF DISPOSAL UNIT CASE STUDY ESTIMATED COST DOCUMENTATION

C.1 Introduction

As part of the overall efforts to evaluate the applicability of CAMU to the proposed Environmental Restoration Disposal Facility at the Hanford Site, a series of 16 case studies were developed and investigated. This investigation included the estimation of disposal unit costs (including trench/vault excavation and construction, treatment, and volume reduction) for each case study. Following are brief discussion presenting the rationale/support for the cost estimates prepared for each case study. Table 1 of the CAMU permit modification outline presents a summary of the waste, design, and treatment alternatives represented by these case studies.

C.2 Trench Construction

Unlined trenches were assumed to be excavated to a total depth of 67 ft with a 1.5:1 side slope. The trench bottom width was assumed to be 100 ft, while the top width was assumed to be 300 ft. These dimensions reflect the values which have been used in other planning documents unrelated to the CAMU evaluation. The main body trench excavation volume was calculated to be 496 yd^3/ft, while the beginning end excavation volume (including corners) was calculated to be 28,800 yd^3. The entry end (i.e., the end at which equipment enters the trench) was assumed to have a side slope of 3:1 to facilitate equipment movement. The entry end excavation volume (including corners) was calculated to be 57,600 yd^3. Excavation unit costs were determined to be \$3.16/yd^3. This unit cost is based on the detailed cost estimates prepared to support overall development and implementation of the large-scale restoration process. The unit cost is based on labor, equipment, supply, and equipment operating costs estimated for each year over a period of 21 yr (using present dollar values).

Lined trenches were assumed to be excavated to a total depth of 38 ft to allow for 3 ft of clay (the synthetic liner base and 2 ft of soil (the soil cover for the synthetic liners). It was assumed that waste would be placed to a depth of 33 ft below grade. Side slopes were determined to be 3:1 on all sides and at both ends of each trench. As with the unlined trenches, these dimensions reflect the values which have been used in other planning documents unrelated to the CAMU evaluation. The lined trench excavation bottom width was determined to be 100 ft while the top width was determined to be 330 ft. The main body trench excavation volume was calculated to be 303 yd^3/ft. The entry and beginning end excavation volume were calculated to be 20,377 yd^3 each. As noted in the unlined trench discussion, the excavation unit cost used in preparing the cost estimates was \$3.16/yd^3.

Trench construction costs also included monitoring well installation costs. For the purposes of these case studies, it was assumes that monitoring wells would be installed in all cases, regardless of the disposal/treatment method. Based on information provided by a WHC geologist familiar with the 200 Area, it was assumed that wells would be installed to a depth of 300 ft. To minimize well installation costs, the trenches would be aligned parallel to the groundwater flow gradient, thereby minimizing the trench aspect intersecting the gradient. The WHC geologist commented that the radius of influence for such monitoring wells in the 200 Area would be 300 to 400 ft. Therefore, the number of wells for one trench (based on the trench dimensions noted above) would total one upgradient well and two downgradient wells. For each additional trench there would be no increase in the number of upgradient wells and an increase of one downgradient well. For example, if there were two trenches there would be one downgradient well at each outer trench corner and one well between the trenches for a total of three downgradient wells. It should be noted that the trenches would be placed within 100 ft of each other. Using information developed by WHC on a number of investigatory efforts throughout the facility the monitoring well installation/ development unit cost is $1,732/ft.

1.3 Liner Systems

The liner system components consisted of a 3-ft-thick clay base, one or two geonet drainage/synthetic membrane layers, and a 2-ft-thick soil cover.

1.3.1 Clay Base

With a thickness of 3 ft, the quantity of clay required at each end of a lined trench was calculated to be 119,790 ft^3. Using the unlined trench dimensions noted above and with a 3-ft- clay thickness, the main trench body clay requirement was calculated to be 1,026 ft^3/ft. Using the commonly accepted engineering value for compacted clay of 120 lb/ft^3, the resulting clay requirements were calculated to be 7,187 tons for each end and 61.56 tons for each linear ft of the main trench segment.

A vendor in the Hanford area provided a quote (reflecting the overall quantities required) of approximately $4.50/ton for clay delivered to the Hanford site in unit trains from a clay mine in Montana. Following are the assumptions used in developing clay handling, transport, and placement estimated costs.

- The clay would be unloaded, by bottom dump hopper cars, at an elevated rail siding in the rail yard of the proposed ERDF. It was assumed that such a rail siding would require approximately one-quarter mile of track. At the standard value of $1,000,000/mile used for laying track at Hanford (based on historical data), the siding would cost $250,000.

- A 7-yd^3 front-end loader would be able to load clay into tractor-trailers (for transport to the excavated trench) at a rate of 677 tons/hr. This value, which would support the projected waste disposal capacity requirements, assumes that a loader with a 7-yd^3 bucket 90% full would operate on a 45-sec cycle for 50 min/hr. The unit cost for this loader (including ownership, operating, and labor costs) is $124.43/hr. Refer to the equipment discussion for additional details on this unit cost. Front end loader hours were based on the number of hours required for tractor-trailer operations.

- The tractor-trailer capacity is the limiting factor, however, for clay transport/placement. With a load of 40 tons per trailer and a 20-min cycle for the rail-trench-rail and unloading round trip, the capacity for each tractor-trailer combination is 120 tons/hr. Tractor-trailer estimated costs were based on using one combination in cases 3, 4, 5, 11, and 12, while (because of its peak requirements) two combinations were assumed in case 9. The unit cost for each tractor-trailer combination was \$87.90/hr. Tractor-trailer hours of operation were determined by dividing clay requirements by the tractor-trailer capacity(ies). Refer to the equipment discussion for additional details on the tractor-trailer unit cost.

- It was assumed that clay placement would require the support of a water truck, sheepsfoot compactor, grader, and vibratory compactor operating the number of hours of the tractor-trailer(s). The unit costs for these items are: \$99.10/hr, \$87.32/hr, \$78.21/hr, and \$87.32/hr, respectively. Additional details on these costs are provided in the equipment discussion.

1.3.2 Geonet-Drainage/Synthetic Membrane Layers

Using the square footages used to calculate clay requirements (clay requirements were calculated by multiplying the square footage of clay base required be the compacted depth of 3 ft). Liner square footage requirements also reflect an overlay of 15 ft (for the anchor trench) around the total perimeter of the trench. Vendor quotes (provided by Gundle Liner Systems) noted an installed cost of approximately \$0.45/ft^2 for the geonet-drainage layer (fabric/net/fabric), and \$0.50/ft for an 80-mil flexible membrane liner (FML). Using commonly accepted engineering values (confirmed in a study on liner systems) it was assumed that inspection, quality assurance oversight, and testing would increase these installed costs by 50%, for resulting unit costs of \$0.68 and \$0.75/ft^2, respectively.

1.3.3 Soil Cover

It was assumed that soils excavated during trench construction would be used to provide a protective cover for the liner. The soil cover requirement was determined by multiplying the synthetic liner square footage requirement by a cover 3 ft thick. Please note, it was assumed that this cover would compact to the desired thickness of 2 ft. The scraper used to move and place the soil is the controlling factor for all other equipment operations. While the scraper could move and place 240 yd^3 of soil per hour (based on the detailed estimates prepared for the large-scale restoration operation and reflecting Hanford-specific operational requirements and historic values from the mining industry), this value was reduced by 50% to ensure care in protecting the synthetic liner. It was assumed that soil cover placement would require the support of a scraper, water truck, 370 hp dozer, grader, and compactor. The unit costs for these items are \$139.36, \$99.10, \$113.13, \$78.21, and \$87.32/hr, respectively.

1.4 EQUIPMENT

Equipment unit costs include ownership, operating, and labor costs. Detailed, bottoms-up estimates for these items were recently prepared to support the development and implementation of the overall large-scale restoration approach at Hanford. Following are the unit costs (based upon averages for all activities over the 21-yr planning period - using present dollars):

• 7-yd^3 front-end loader	$124.43/hr
• 3.5-yd^3 front-end loader	101.38/hr
• water truck	99.10/hr
• grader	78.21/hr
• compactor (sheepsfoot and vibratory)	87.32/hr
• scraper	139.36/hr
• 370-hp dozer	113.13/hr
• tractor/trailer	87.90/hr
• 10-ton dump truck	62.42/hr

1.5 HANFORD BARRIER

The Hanford Barrier was assumed to cover the entire surface area (at grade) occupied by each trench with an overlay of 10 ft around the entire perimeter of the trench. The WHC-developed unit cost for the Hanford Barrier (based on current WHC estimates) was provided as the basis for estimating case study cost. The WHC unit cost is $13.70/$ft^2$.

1.6 RCRA BARRIER

The estimated unit cost for this barrier was developed using the following components: clay, FML, geofabric, filter sand, filter/drainage gravel, biotic barrier, and vegetation. As with the Hanford Barrier, the RCRA Barrier was assumed to cover the entire surface area (at grade) of the trench and vault with the addition of a 10-ft overlap around the entire perimeter of the trench and vault.

1.6.1 Clay

For the purposes of this barrier, clay would be placed to a thickness of 2 ft over the entire barrier surface area. After calculating clay requirements, using the volume and to conversions presented above in the liner discussion, the same equipment costs and unit hours requirements described above were applied to the clay quantities. It should be noted that estimated costs for a rail siding were not added to the trench barrier determinations as rail siding costs were already included in the liner costs. All trenches requiring a RCRA Barrier also have a liner. As the vault does not incorporate a liner system, costs for a rail siding were added to barrier costs in this particular instance.

1.6.2 Flexible Membrane Liner

The unit cost for this item was based upon a vendor quote (from Gundle) of \$0.40/ft^2 for an installed 60-mil FML. Using the 50% oversight/testing factor described above in the liner discussion, the resulting FML estimated unit cost is \$0.60/ft^2.

1.6.3 Geofabric

A vendor (Gundle) provided a quote of \$0.20/ft^2 for an installed geofabric layer. Because the geofabric does not have to be welded/sealed along its seams, the oversight/testing factor was reduced to 10%, for a resulting unit cost of \$0.22/ft^2. It should be noted that the RCRA Barrier has two geofabric layers.

1.6.4 Filter Sand, Filter/Drainage Gravel, Biotic Barrier, and Vegetation

Unit costs for the above items were derived from DOE report DOE/RL-93-35 (a design report for ERDF construction). The filter sand and filter/drainage gravel layers were assumed to be 0.5 ft thick. The biotic barrier consisted of an 8-in. layer of crushed basalt and a 4-in. layer of gravel. The resulting unit costs for these items follow:

• filter sand	\$0.52/ft^2
• filter/drainage gravel	0.37/ft^2
• biotic barrier	0.50/ft^2
• vegetation	0.08/ft^2

1.7 STABILIZATION

Conversations with Chemical Waste Management and Laidlaw marketing staff, and, particularly, with the site manager for the Chemical Waste Management Arlington, Oregon, treatment and disposal facility, it was learned that typical unit costs (including equipment, labor, materials, and testing) for stabilization ranged from \$40/ton to \$80/ton depending on the type of stabilization material used (i.e., fly ash or Portland cement). For the purpose of these case study estimates, the mid-range value of \$60/ton was used. Using an excavated waste density of 2,640 lb/yd^3 yields a unit cost of \$79/yd^3.

It should be noted that the trench capacities for stabilized waste reflect a volume increase of 50% for the waste stabilized. This increase reflects the typical addition of

0.4 ton of stabilizing material for each ton of waste (assuming the material is excavated soil, as opposed to sludge) and was confirmed by the Chemical Waste Management site manager noted above.

1.8 VITRIFICATION

Information provided in two papers prepared by an employee of FERMCO was used to develop the estimated unit cost for vitrification. It should be noted that the information provided in these papers showed the combined effects of a number of operations, therefore, it was necessary to segregate those costs which were specific to vitrification (i.e., did not include redial investigation/feasibility study performance or other treatment or volume reduction operations).

Values from the March 1992 paper showed a total cost of $231,525,000 for the treatment of 460,000 m^3 of waste.

Values presented in the April 1993 paper showed a total of $63,000 for vitrifying 60 m^3 of waste (50 m^3 of sludge plus 50 m^3 of soils reduced, via washing, to 10 m^3).

The average of these two values is $776/$m^3$ or $592/$yd^3$. For the purpose of the case studies, this value was rounded to $600/$yd^3$.

It should be noted that, based on information provided in these papers, vitrification yields a 15% reduction in waste volume. This reduction was used in determining waste disposal volume requirements.

1.9 VOLUME REDUCTION

For the purpose of these case studies, it was assumed that volume reduction would reduce the waste volume by 80% and that the all contaminants would be contained in the remaining 20%. Information provided by Scientific Ecology Group, a vendor of soil washing systems, indicated that a 100-ton/hr system would provide useful reference cost information. The representative commented that, depending on the level of treatment provided, this system would operate with a unit cost of $50 to $200/ton. As these case studies are not attempting to provide treatment per se, and operating simply to achieve a gross volume reduction, the case study estimates used the value of $50/ton ($66/yd^3).

The trenches and vaults that received volume-reduced wastes were sized to reflect the 80% volume reduction noted above.

1.10 VAULT CONSTRUCTION

Vault dimensions were based on the values developed in previous ERDF and large-scale restoration efforts. The vault interior dimensions are 24 ft deep by 150 ft wide. The reinforced-concrete slab was assumed to be 2 ft thick, while the walls were assumed to be 1-ft thick. Vault construction estimated costs consisted of excavation, concrete construction, and side backfill costs.

1.10.1 Vault Excavation

Excavation volumes were calculated using 1.5:1 side slopes for the sides and the beginning end of the excavation, and a 3:1 slope for the entry end (to allow equipment entry) of the excavation. As with trench excavation, the estimated unit cost for excavation was determined to be $3.16/yd^3. The total excavation volume was calculated to be 4,751,440 yd^3 for a total estimated cost of $15,014,550.

1.10.2 Concrete Construction

Reinforced-concrete quantities were calculated using the dimensions noted above and the linear foot values noted on the case study summaries. Standard Hanford unit prices for formed, reinforced concrete (including all contractor charges) are $100/yd^3 for slabs and $200/yd^3 for walls. For the purpose of the case studies, the slab value was increased by 10% to account for forming sumps and sloping the slab to the sumps (to collect any liquids which may be present during construction or waste placement). The total concrete cost was calculated to be $46,667,000.

To provide an additional barrier, it was assumes that the concrete interior surface would be covered with an asphaltic type of coating. The total unit cost, including the local factor, provided in the Means cost estimating guide for this coating is $1.92/SF. The resulting total coating cost is $10,741,000.

1.10.3 Backfill

Backfill costs were developed using unit costs derived from the detailed cost estimates prepared recently for the large-scale restoration study. These unit costs include all labor, equipment, and supply charges. The soils removed during excavation would be used for backfill. Reflecting the varying abilities to work in and place equipment in a sloping excavation, which is very narrow at its base, unit costs and total costs were developed for the first 3 ft of backfill, for the next 7 ft, and for the remainder. Hand equipment was used as the basis for the first value, while the last value assumed the use of a compactor. The respective estimated unit costs are $28.59/yd^3, $29.47/yd^3, and $2.49/yd^3. The total estimated cost for backfill was determined to be $6,413,000.

1.11 100-YEAR MAINTENANCE COSTS

These costs include sampling and analysis and maintenance. Sampling and analysis costs were based on quarterly sampling for the first 5 yr followed by annual sampling thereafter. Analytical charges for each sample (reflecting the use of a somewhat reduced list of analytes) were estimated to be $2,000 per sample. Sample collection costs were based on the use of a two-person contractor crew working 3 hr per well and charging $75/hr/person. The total estimated unit cost of $2,450 (analysis and collection) was rounded to $2,500 for the purpose of this estimate.

Maintenance costs were estimated on the basis of one crew working one 8-hr shift per day, 5 days/week for 1 month (22 working days) each year. The work crew was based on one supervisor, six laborers, and one health physics technician. The total unit cost for this crew (based on current average WHC rates) is $1,647.94/shift.

It was also assumed that the work crew would need a 370-hp dozer, compactor, grader, 3.5-yd^3 loader, and 10-ton dump truck to support their cover and facility maintenance activities. The unit costs for these items are provided above in the equipment discussion. Reflecting current and historic Hanford conditions, the equipment would be used 5 hr per shift (to accommodate lunch, breaks, decon, etc.). The total estimated costs for equipment were based on this length of operation.

CASE STUDY COST ESTIMATE OBSERVATIONS

Following is a summary of unit costs for the various disposal/treatment/volume reduction/barrier combinations evaluated in the case studies (these estimated unit costs are based on averages where more than one case used the noted combination).

• unlined trench/no barrier	\$ 5.89/yd^3
• unlined trench/Hanford Barrier	14.11/yd^3
• single liner system with a Hanford Barrier	28.84/yd^3
• single liner system with a RCRA Barrier	14.91/yd^3
• MTR liner system with a RCRA Barrier	16.65/yd^3
• volume reduction with stabilization	81.81/yd^3
• volume reduction with vitrification	186.05/yd^3
• volume reduction	66.00/yd^3
• stabilization	79.00/yd^3
• vitrification	600.00/yd^3

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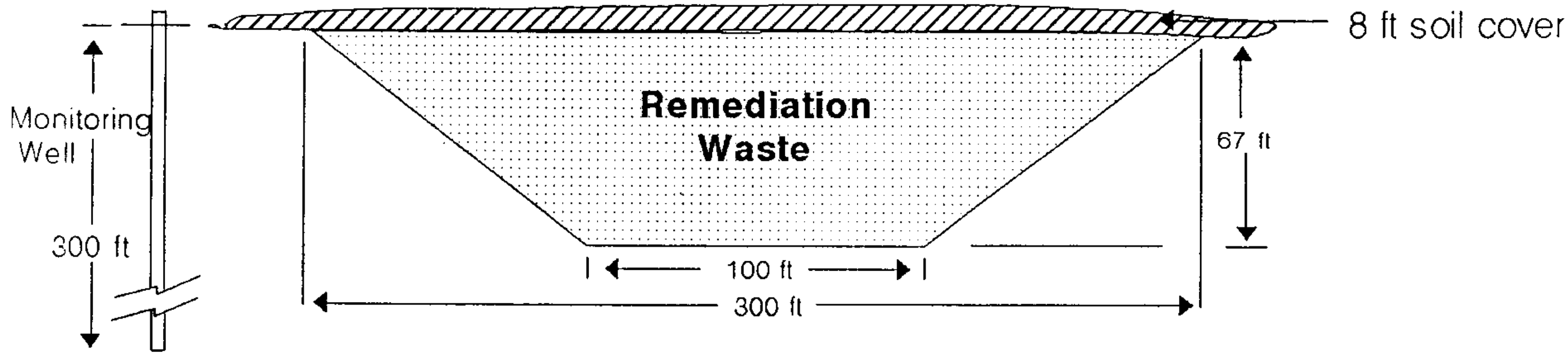
WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 1



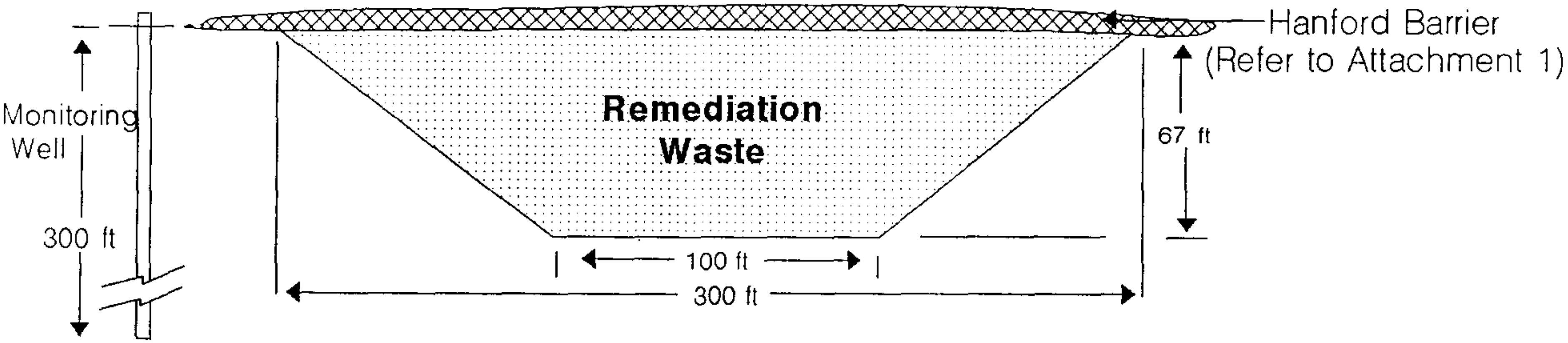
CASE STUDY NO. 1 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	6,021,000	4.26
Volume Reduction	0	0.00
Liner System	0	0.00
Barrier (Soil Cover)	1,376,000	0.97
Waste Form	0	0.00
100-Year Maintenance Cost	10,028,000	7.10
Total	17,425,000	12.34

CASE STUDY NO. 1 INFORMATION	
Remediation Waste Codes(s)	D and E
Remediation Waste Quantity [1]	1,412,000 CY
Trench Length	2,973 LF
Trench Capacity	1,412,200 CY

[1] Based on 1992 ER estimates

Total Estimated Disposal Unit Costs for Waste Codes D and E are $17,425,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 2



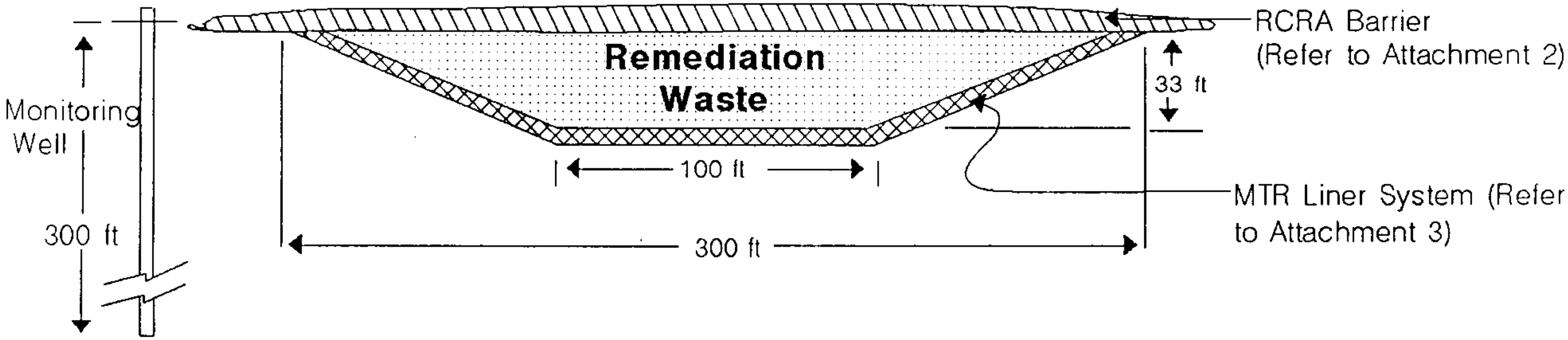
CASE STUDY NO. 2 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	6,021,000	4.26
Volume Reduction	0	0.00
Liner System	0	0.00
Barrier (Hanford)	13,121,000	9.29
Waste Form	0	0.00
100-Year Maintenance Cost	10,028,000	7.10
Total	29,170,000	20.66

CASE STUDY NO. 2 INFORMATION	
Remediation Waste Codes(s)	D and E
Remediation Waste Quantity [1]	1,412,000 CY
Trench Length	2,973 LF
Trench Capacity	1,412,200 CY

[1] Based on 1992 ER estimates

Total Estimated Disposal Unit Costs for Waste Codes D and E are $29,170,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 3



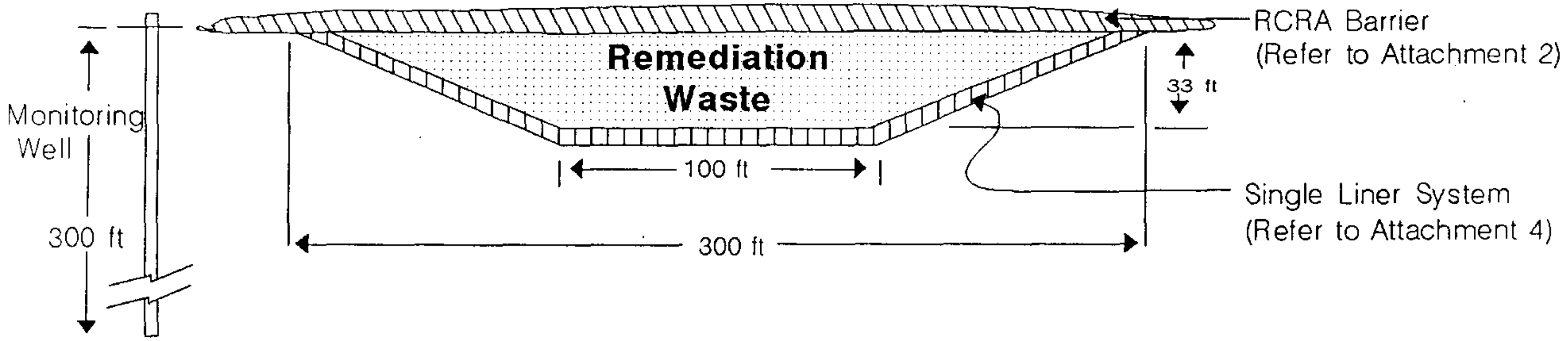
CASE STUDY NO. 3 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	7,095,000	5.02
Volume Reduction	0	0.00
Liner System (Min. Tech. Requirements)	10,151,000	7.19
Barrier (RCRA)	6,797,000	4.81
Waste Form	0	0.00
100-Year Maintenance Cost	10,028,000	7.10
Total	34,071,000	24.13

CASE STUDY NO. 3 INFORMATION	
Remediation Waste Codes(s)	D and E
Remediation Waste Quantity [1]	1,412,000 CY
Trench Length	5,848 LF
Trench Capacity	1,412,160 CY

[1] Based on 1992 ER estimates

Total Estimated Disposal Unit Costs for Waste Codes D and E are $34,071,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 4



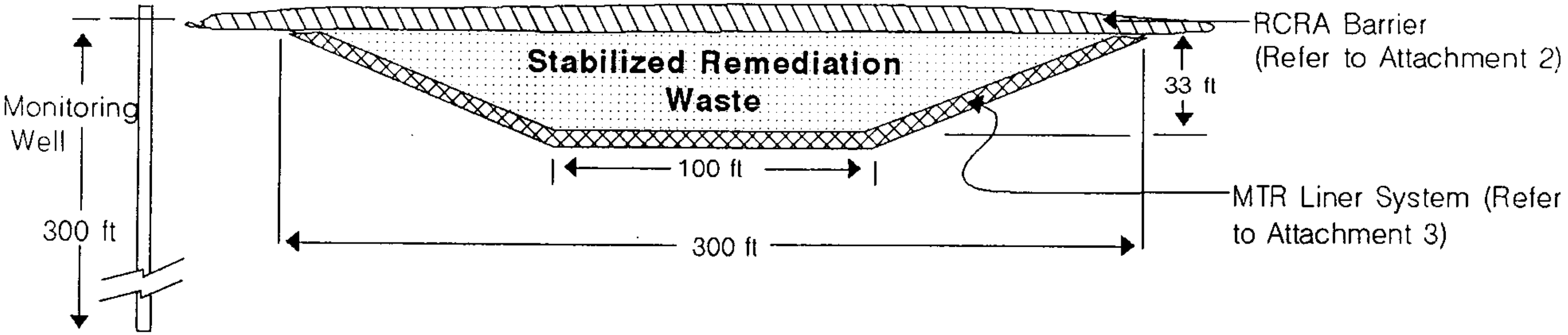
CASE STUDY NO. 4 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	7,095,000	5.02
Volume Reduction	0	0.00
Liner System (Single)	7,170,000	5.08
Barrier (RCRA)	6,797,000	4.81
Waste Form	0	0.00
100-Year Maintenance Cost	10,028,000	7.10
Total	31,090,000	22.02

CASE STUDY NO. 4 INFORMATION	
Remediation Waste Codes(s)	D and E
Remediation Waste Quantity [1]	1,412,000 CY
Trench Length	5,848 LF
Trench Capacity	1,412,160 CY

[1] Based on 1992 ER estimates

Total Estimated Disposal Unit Costs for Waste Codes D and E are $31,090,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 5



CASE STUDY NO. 5 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	10,392,000	7.36
Volume Reduction	0	0.00
Liner System (Min. Tech. Requirements)	15,204,000	10.77
Barrier (RCRA)	10,256,000	7.26
Waste Form (Stabilized)	111,548,000	79.00
100-Year Maintenance Cost	10,315,000	7.31
Total	157,715,000	111.70

CASE STUDY NO. 5 INFORMATION	
Remediation Waste Codes(s)	D and E
Remediation Waste Quantity [1]	1,412,000 CY (2,118,000 CY disposed [2])
Trench Length	2 @ 4,407 LF
Trench Capacity	2,118,200 CY

[1] Based on 1992 ER estimates

[2] Reflects 50% volume increase from stabilization.

Total Estimated Disposal Unit Costs for Waste Codes D and E are $<u>157,715,000</u>.

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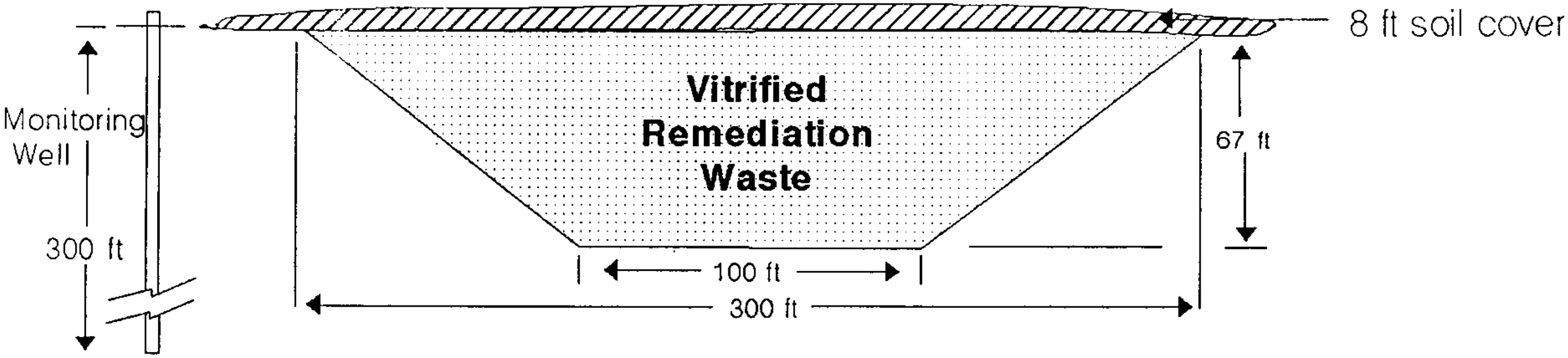
WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 6



CASE STUDY NO. 6 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	5,352,000	3.79
Volume Reduction	0	0.00
Liner System	0	0.00
Barrier (Soil Cover)	1,133,000	0.80
Waste Form (Vitrified)	847,200,000	600.00
100-Year Maintenance Cost	10,028,000	7.10
Total	863,713,000	611.69

CASE STUDY NO. 6 INFORMATION	
Remediation Waste Codes(s)	D and E
Remediation Waste Quantity [1]	1,412,000 CY (1,200,000 CY as disposed [2])
Trench Length	2,446 LF
Trench Capacity	1,200,400 CY

[1] Based on 1992 ER estimates

[2] This quantity reflects the 15% volume reduction following vitrification

Total Estimated Disposal Unit Costs for Waste Codes D and E are $863,713,000.

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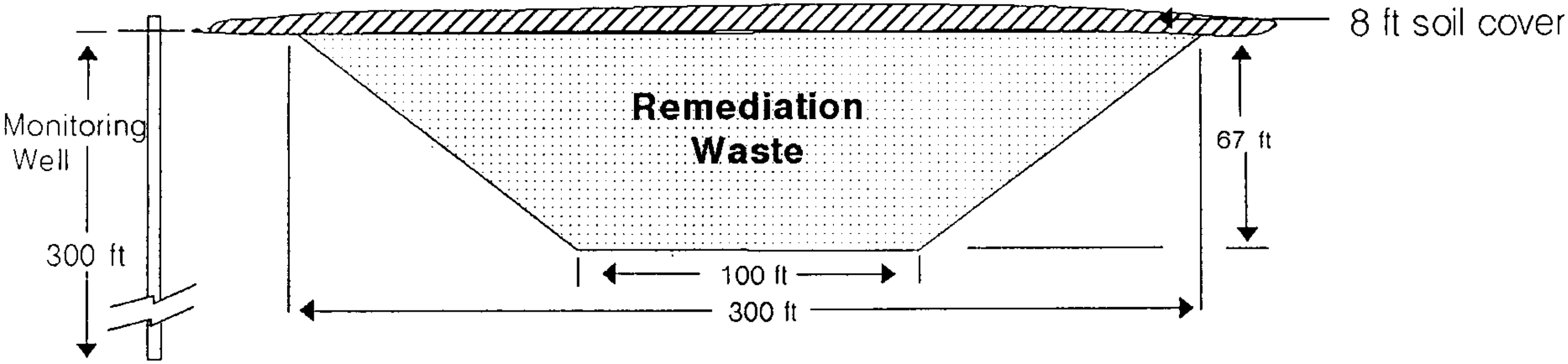
WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 7



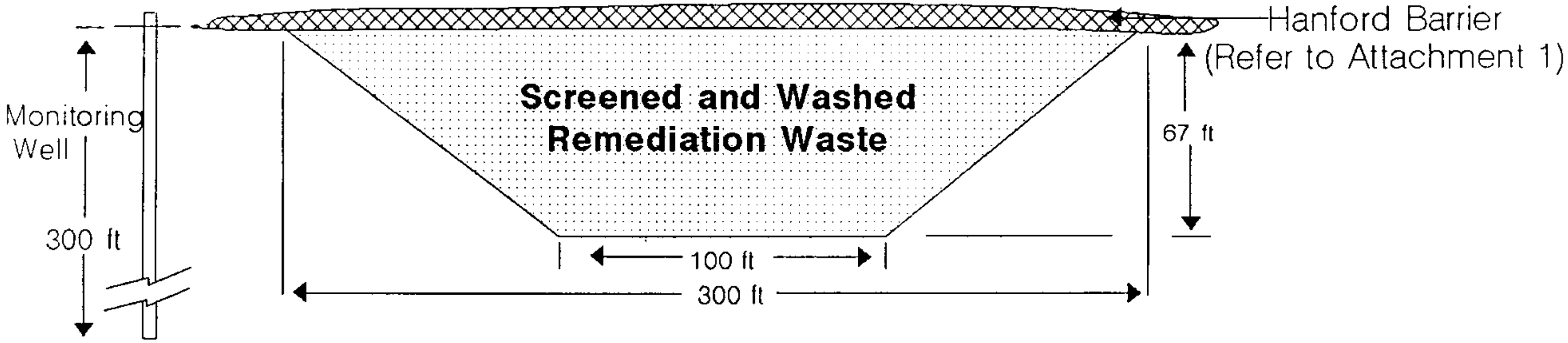
CASE STUDY NO. 7 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	9,800,000	3.76
Volume Reduction	0	0.00
Liner System	0	0.00
Barrier (Soil Cover)	2,484,000	0.95
Waste Form	0	0.00
100-Year Maintenance Cost	10,028,000	3.85
Total	22,312,000	8.56

CASE STUDY NO. 7 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY
Trench Length	5,384 LF
Trench Capacity	2,608,100 CY

[1] Based on 1992 ER estimates

Total Estimated Disposal Unit Costs for Waste Code C are $22,312,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 8



CASE STUDY NO. 8 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	3,210,000	1.23
Volume Reduction (Screened and Washed)	172,128,000	66.00
Liner System	0	0.00
Barrier (Hanford)	5,257,000	2.02
Waste Form	0	0.00
100-Year Maintenance Cost	10,028,000	3.85
Total	190,623,000	73.09

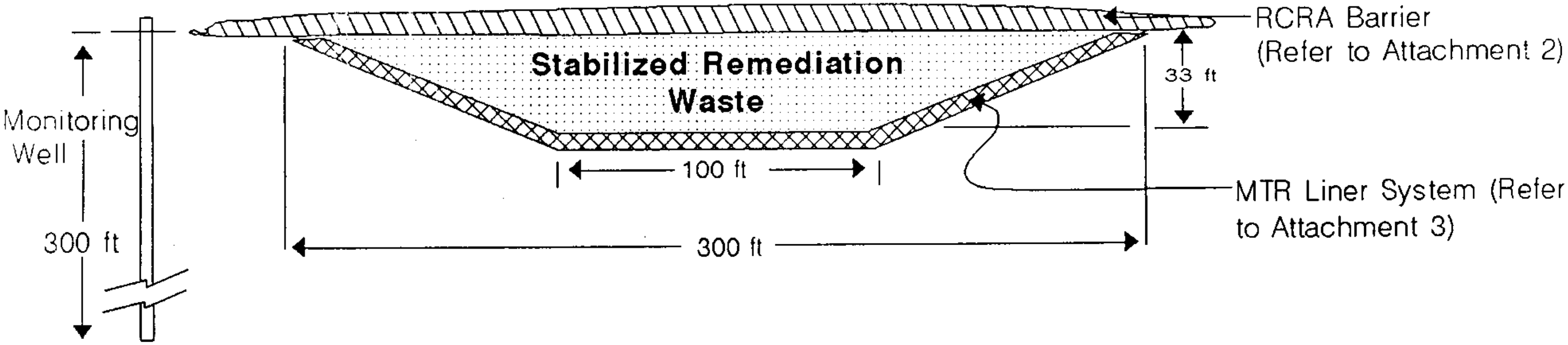
CASE STUDY NO. 8 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY (522,000 CY after volume reduction [2])
Trench Length	1,179 LF
Trench Capacity	522,400 CY

[1] Based on 1992 ER estimates

[2] This quantity reflects the 80% volume reduction achieved by screening and washing.

Total Estimated Disposal Unit Costs for Waste Code C are $190,623,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 9



CASE STUDY NO. 9 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	17,941,000	6.88
Volume Reduction	0	0.00
Liner System (MTR)	25,734,000	9.87
Barrier (RCRA)	18,859,000	7.23
Waste Form (Stabilized)	206,032,000	79.00
100-Year Maintenance Cost	10,603,000	4.07
Total	279,169,000	107.04

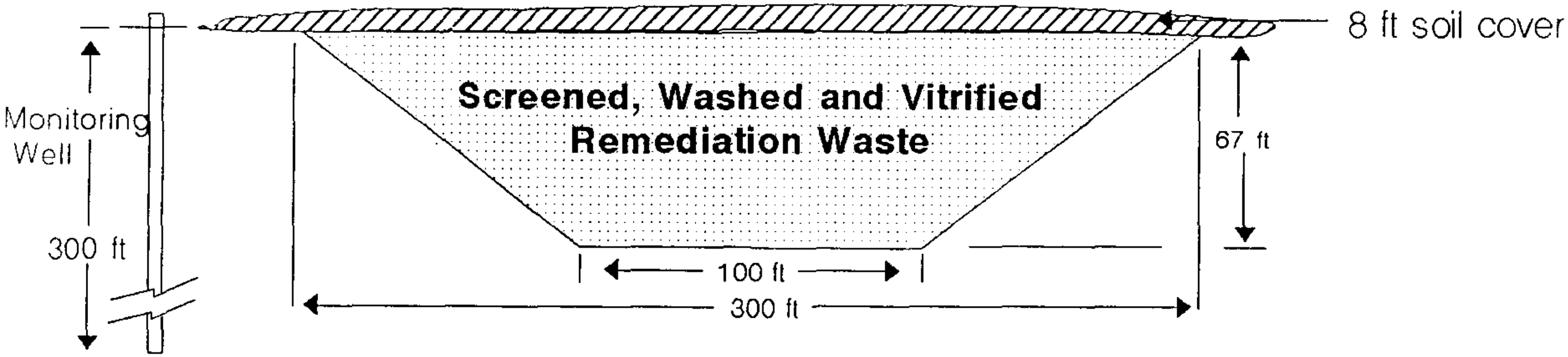
CASE STUDY NO. 9 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY (3,912,000 CY Disposed [2])
Trench Length	3 @ 5,407 LF
Trench Capacity	3,912,300 CY

[1] Based on 1992 ER estimates

[2] Reflects the 50% volume increase from stabilization.

Total Estimated Disposal Unit Costs for Waste Code C are $279,169,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 10



CASE STUDY NO. 10 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	2,962,000	1.14
Volume Reduction (Screened and Washed)	172,128,000	66.00
Liner System	0	0.00
Barrier (Soil Cover)	432,000	0.17
Waste Form (Vitrified)	313,200,000	120.09
100-Year Maintenance Cost	10,028,000	3.85
Total	498,750,000	191.24

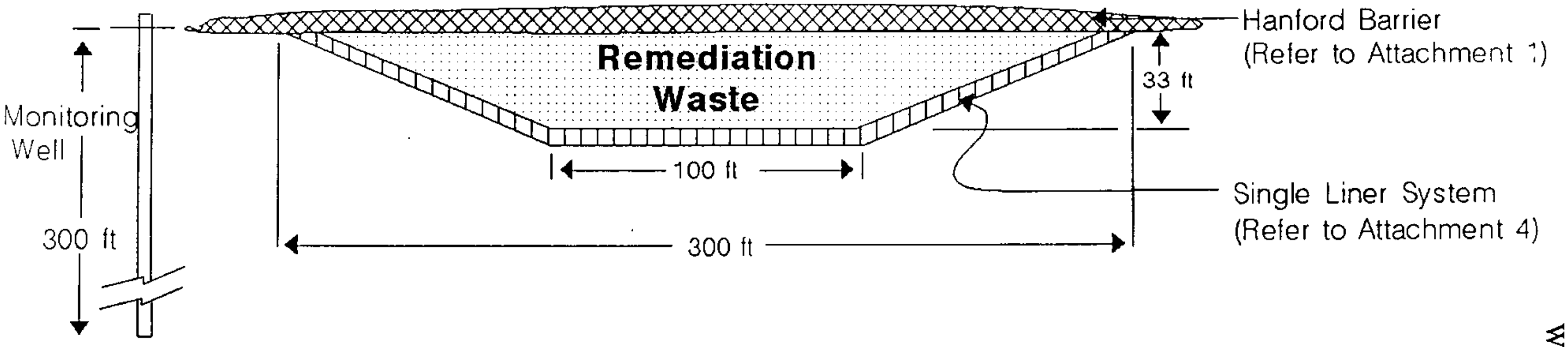
CASE STUDY NO. 10 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY (444,000 CY Disposed[2])
Trench Length	921 LF
Trench Capacity	444,000 CY

[1] Based on 1992 ER estimates

[2] This quantity reflects the 80% volume reduction achieved by screening and washing and the 15% volume reduction following vitrification

Total Estimated Disposal Unit Costs for Waste Code C are $498,750,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 11



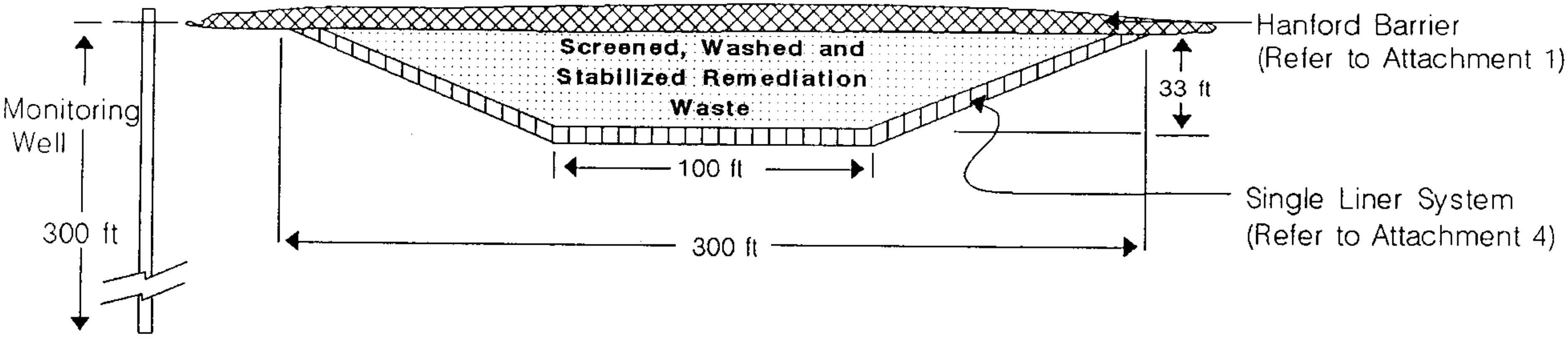
CASE STUDY NO. 11 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	12,307,000	4.72
Volume Reduction	0	0.00
Liner System (Single)	13,052,000	5.00
Barrier (Hanford)	47,584,000	18.25
Waste Form	0	0.00
100-Year Maintenance Cost	10,315,000	3.96
Total	83,258,000	31.92

CASE STUDY NO. 11 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY
Trench Length	2 @ 5,407 LF
Trench Capacity	2,608,200 CY

[1] Based on 1992 ER estimates

Total Estimated Disposal Unit Costs for Waste Code C are $83,258,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 12



CASE STUDY NO. 12 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	4,637,000	1.78
Volume Reduction (Screened and Washed)	172,128,000	66.00
Liner System (Single)	4,150,000	1.59
Barrier (Hanford)	14,467,000	5.55
Waste Form (Stabilized)	41,238,000	15.81
100-Year Maintenance Cost	10,028,000	3.85
Total	246,648,000	94.57

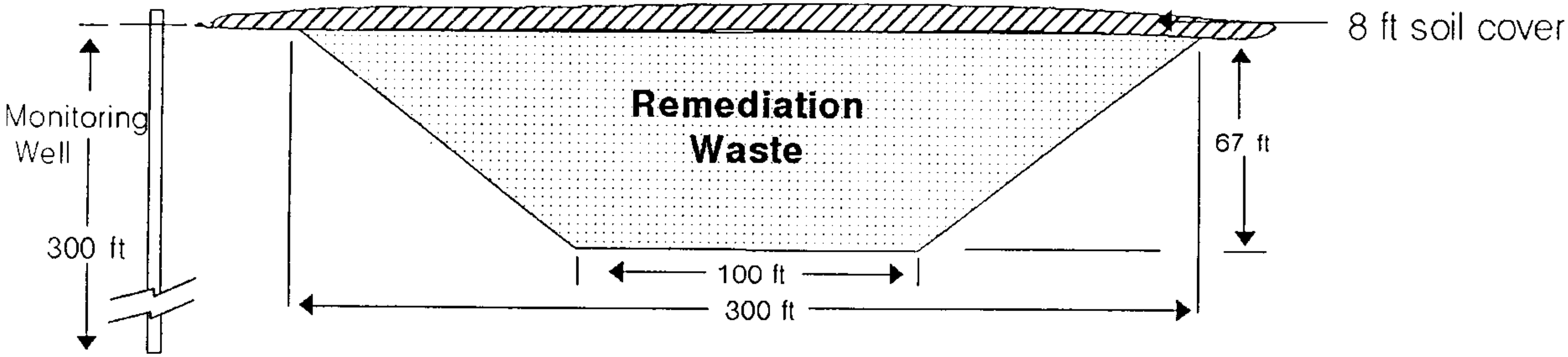
CASE STUDY NO. 12 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY (783,000 CY Disposed[2])
Trench Length	3,280 LF
Trench Capacity	783,000 CY

[1] Based on 1992 ER estimates

[2] This quantity reflects the 80% volume reduction achieved by screening and washing, and the 50% volume increase following stabilization.

Total Estimated Disposal Unit Costs for Waste Code C are $246,648,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 13



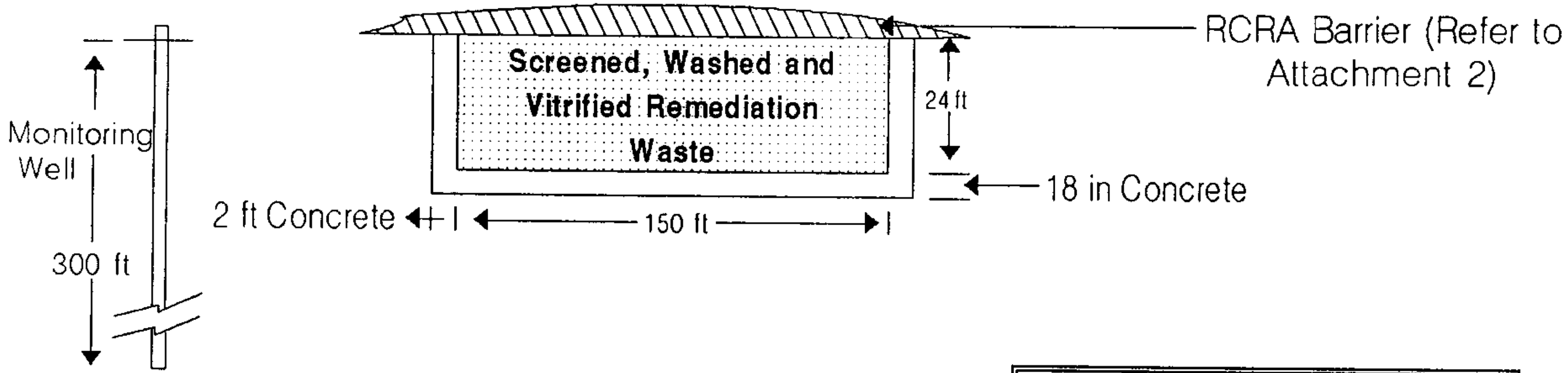
CASE STUDY NO. 13 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	65,808,000	3.34
Volume Reduction	0	0.00
Liner System	0	0.00
Barrier (Soil Cover)	18,568,000	0.94
Waste Form	0	0.00
100-Year Maintenance Cost	11,178,000	0.57
Total	95,554,000	4.86

CASE STUDY NO. 13 INFORMATION	
Remediation Waste Codes(s)	A and B
Remediation Waste Quantity [1]	19,674,000 CY
Trench Length	5 @ 8,059 LF
Trench Capacity	19,674,300 CY

[1] Based on 1992 ER estimates

Total Estimated Disposal Unit Costs for Waste Codes A and B are $95,554,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 14



CASE STUDY NO. 14 ESTIMATED COSTS		
COMPONENT	COST PER VAULT	UNIT COST ($/CY)
Vault Excavation/ Construction	82,483,000	4.19
Volume Reduction (Screened and Washed)	1,298,484,000	66.00
Liner System	0	0.00
Barrier (RCRA)	15,667,000	0.80
Waste Form (Furnace Vitrified)	2,361,000,000	120.01
100-Year Maintenance Cost	11,178,000	0.57
Total	3,768,812,000	191.56

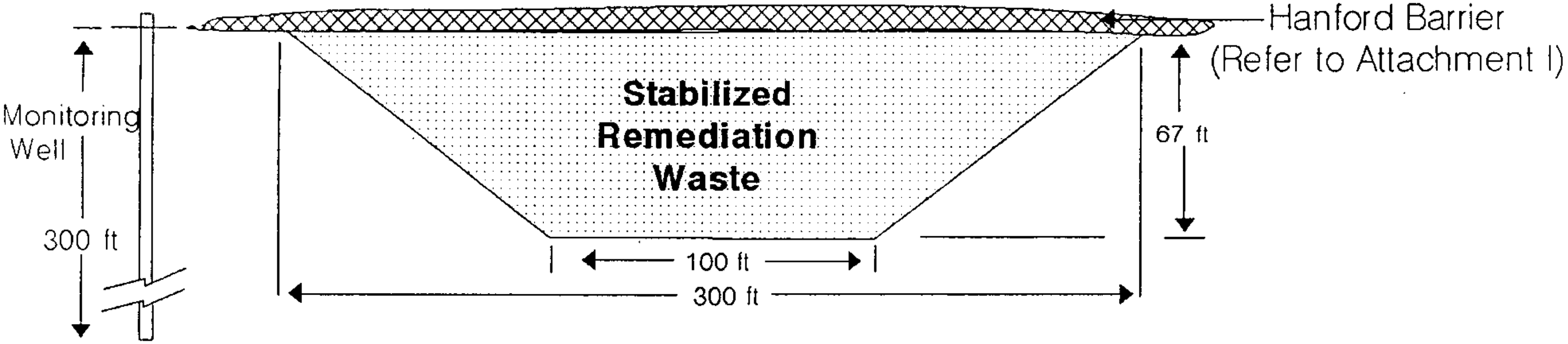
CASE STUDY NO. 14 INFORMATION	
Remediation Waste Codes(s)	A and B
Remediation Waste Quantity [1]	19,674,000 CY (3,345,000 CY Disposed [2])
Vault Length	4 @ 5,132 LF 1 @ 4,830 LF
Vault Capacity	3,352,000 CY

[1] Based on 1992 ER estimates

[2] This quantity reflects the 80% volume reduction achieved by screening and washing and the 15% volume reduction following vitrification

Total Estimated Disposal Unit Costs for Waste Codes A and B are $3,768,812,000.

WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 15



CASE STUDY NO. 15 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	13,921,000	5.34
Volume Reduction	0	0.00
Liner System	0	0.00
Barrier (Hanford)	35,217,000	13.50
Waste Form (Stabilized)	206,032,000	79.00
100-Year Maintenance Cost	10,028,000	3.85
Total	265,198,000	101.69

CASE STUDY NO. 15 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY (3,912,000 CY disposed [2])
Trench Length	8,013 LF
Trench Capacity	3,912,100 CY

[1] Based on 1992 ER estimates

[2] Reflects the 50% volume increase from stabilization.

Total Estimated Disposal Unit Costs for Waste Code C are $265,198,000.

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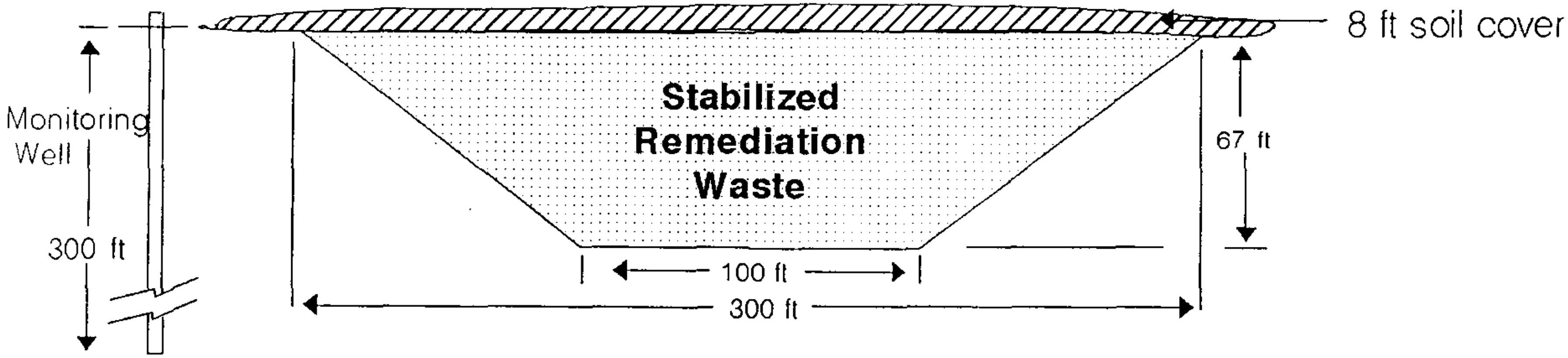
WHC ERSDF DISPOSAL UNIT CASE STUDY DESCRIPTION AND ESTIMATED COSTS - CASE STUDY NO. 16



CASE STUDY NO. 16 ESTIMATED COSTS		
COMPONENT	COST PER TRENCH	UNIT COST ($/CY)
Trench Excavation/ Construction	13,921,000	5.34
Volume Reduction	0	0.00
Liner System	0	0.00
Barrier (Soil Cover)	3,692,000	1.42
Waste Form (Stabilized)	206,032,000	79.00
100-Year Maintenance Cost	10,028,000	3.85
Total	233,673,000	89.60

CASE STUDY NO. 16 INFORMATION	
Remediation Waste Codes(s)	C
Remediation Waste Quantity [1]	2,608,000 CY (3,912,000 CY disposed [2])
Trench Length	8,013 LF
Trench Capacity	3,912,100 CY

[1] Based on 1992 ER estimates

[2] Reflects the 50% volume increase from stabilization.

Total Estimated Disposal Unit Costs for Waste Code C are $233,673,000.