DISCLAIMER

This report was prepared as an account of work sponsored by an agency of the United States Government. Neither the United States Government nor any agency thereof, nor any of their employees, makes any warranty, express or implied, or assumes any legal liability or responsibility for the accuracy, completeness, or usefulness of any information, apparatus, product, or process disclosed, or represents that its use would not infringe privately owned rights. Reference herein to any specific commercial product, process, or service by trade name, trademark, manufacturer, or otherwise does not necessarily constitute or imply its endorsement, recommendation, or favoring by the United States Government or any agency thereof. The views and opinions of authors expressed herein do not necessarily state or reflect those of the United States Government or any agency thereof.

INEL-95/0095
(formerly EGG-WM-11415)
Revision 1

Preliminary Parametric Performance Assessment of Potential Final Waste Forms for Alpha Low-Level Waste at the Idaho National Engineering Laboratory

**T. H. Smith
M. E. Sussman
Lockheed Idaho Technologies Company**

**J. Myers
S. M. Djordjevic
T. A. DeBiase
M. T. Goodrich
D. DeWitt
IT Corporation
Albuquerque, New Mexico**

Published August 1995

**Idaho National Engineering Laboratory
Lockheed Idaho Technologies Company
Idaho Falls, Idaho 83415**

**Prepared for the
U.S. Department of Energy
Assistant Secretary for Environmental Management
Under DOE Idaho Operations Office
Contract DE-AC07-94ID13223**

MASTER

DISTRIBUTION OF THIS DOCUMENT IS UNLIMITED

the

DISCLAIMER

Portions of this document may be illegible in electronic image products. Images are produced from the best available original document.

ABSTRACT

This report presents a preliminary parametric performance assessment (PA) of potential waste disposal systems for alpha-contaminated, mixed, low-level waste (ALLW) currently stored at the Transuranic Storage Area of the Department of Energy's (DOE's) Idaho National Engineering Laboratory. The ALLW, which contains from 10 to 100 nCi/g of transuranic (TRU) radionuclides, is awaiting treatment and disposal.

The purpose of this study was to examine the effects of several parameters on the radiological-confinement performance (in terms of projected receptor doses) of potential disposal systems for the ALLW. The principal emphasis was on the performance of final waste forms (FWFs). Three categories of FWF (cement, glass, and ceramic) were addressed by evaluating the performance of two limiting FWFs for each category. Performance at five conceptual disposal sites was evaluated to illustrate the effects of site characteristics on the performance of the total disposal system. Other parameters investigated for effects on receptor dose included inventory assumptions, TRU radionuclide concentration, FWF fracture, disposal depth, water infiltration rates, subsurface-transport modeling assumptions, receptor well location, intrusion scenario assumptions, and the absence of waste immobilization. These and other factors were varied singly and in some combinations.

The results indicate that compliance of the treated and disposed ALLW with the performance objectives given in DOE Order 5820.2A, Chapter III, depends on the assumptions made, as well as on the FWF and the disposal site. Some combinations result in compliance, while others do not. The implications of these results for decisionmaking relative to treatment and disposal of the INEL ALLW are discussed.

The report compares the degree of conservatism in this preliminary parametric PA against that in four other PAs and one risk assessment. All of the assessments addressed the same disposal site, but different wastes.

The report also presents a qualitative evaluation of the uncertainties in the PA and makes recommendations for further study.

EXECUTIVE SUMMARY

Introduction and Background

This report presents a preliminary parametric performance assessment (PA) of potential waste disposal systems for alpha-contaminated, mixed, low-level waste (ALLW) currently stored at the Transuranic Storage Area (TSA) of the Department of Energy's (DOE's) Idaho National Engineering Laboratory (INEL). The ALLW contains from 10 to 100 nCi/g of transuranic (TRU) radionuclides and was generated primarily from 1970 through 1989.

The containers of ALLW are stored together with similar containers of TRU waste having >100 nCi/g of TRU contamination. The volume of ALLW is estimated to be $24,413 \text{ m}^3$, about 40% of the total volume of $63,729 \text{ m}^3$ of the waste stored at the TSA (ALLW plus TRU waste). The waste is awaiting retrieval from storage, segregation into TRU waste versus ALLW, treatment, and disposal.

This PA was conducted to help evaluate potential treatment and disposal of the ALLW. The treatment evaluations are in their early stages. Feasibility studies on private sector treatment have been performed, but a final waste form (FWF) and a disposal site for the ALLW have not yet been selected.

The purpose of this study was to provide preliminary evaluations of the radionuclide-confinement performance of treated ALLW in various FWFs, in various disposal locations, and under a variety of assumptions. The study examined the effects of several parameters on the performance of the overall disposal system (consisting of the FWF, container, engineered barriers, and disposal site). The performance was evaluated in terms of the projected receptor doses and the other performance objectives of DOE Order 5820.2A, Chapter III.

The principal emphasis was on the performance of potential FWFs for the ALLW. Three broad categories of FWF (cement, glass, and ceramic) were addressed by evaluating the performance of two limiting FWFs for each category.

Performance at five possible disposal sites was evaluated to illustrate the effects site characteristics would have on the performance of the total disposal system. Several other parameters were investigated, both singly and in some combinations, to determine the effects on the calculated receptor dose. Not every combination of parameters was evaluated, however.

This study is a preliminary parametric PA. It is preliminary in that it is not the formal PA that will be prepared after a disposal site and the FWF(s) have been selected. It is parametric in that several parameters were varied to observe their effects on the results, rather than examining in detail one FWF and one disposal site. At this early time in the treatment-selection process, the evaluation of a wide range of options was needed to provide guidance on the selection of a FWF(s).

The study was performed in three phases, all of which are reported herein. In the first phase, PA computer runs were performed using an initial set of assumptions and parameter

values. In the second and third phases, the earlier results were evaluated, then additional runs were made using refined assumptions or expanded ranges for some parameter values. The complete set of earlier runs was not repeated using the revised assumptions or parameter values.

This study has several potential uses:

- It will help in determining the optimum degree of ALLW immobilization to satisfy the performance objectives of DOE Order 5820.2A, while minimizing cost
- The results indicate the margin of compliance with the performance objectives and how the margin depends on various parameters
- The study identifies key parameters for which better data are needed.

The Waste

Most of the INEL ALLW was generated at the Rocky Flats Plant; the remainder, at several other DOE facilities, including the INEL. The waste consists of a wide variety of solid materials (e.g., sludges, paper, metals) contaminated with small amounts of TRU radionuclides and other actinides. Some of the ALLW, particularly the small fraction that is remote handled, also contains fission products and activated metals.

Most of the ALLW is mixed waste; i.e., it contains chemically hazardous constituents per the Resource Conservation and Recovery Act (RCRA). Treatment will be required under the Land Disposal Restrictions of RCRA.

Shipping records for the waste are incomplete, particularly concerning radionuclide information. As a result of this situation and the uncertain split of the waste between ALLW and TRU waste, there is some uncertainty about the inventory of radionuclides in the ALLW. Based on a set of assumptions, a simplified, limiting inventory was developed for initial use in the PA. For the second phase, a refined inventory for part of the waste was used in some computer runs. For the third phase, an inventory with additional refinements for part of the waste was used in additional runs.

Glass, cement, and ceramic were considered logical candidates for the FWF that will be chosen for the ALLW. To evaluate a wide range of potential variations within these three categories, two limiting FWFs were selected for evaluation from each category. Of each pair, one represents the more favorable characteristics in terms of contaminant immobilization; the other represents the less favorable characteristics. Table ES-1 lists the FWFs evaluated.

Disposal Sites

The disposal location for the treated ALLW has not been selected. To illustrate the effects of disposal site characteristics on the performance of the disposal system, five sites were used. Characteristics that are representative of the following sites were chosen: the Subsurface Disposal Area at the INEL, Site 14 in the central part of the INEL, the Radioactive Waste

Table ES-1. Final waste forms selected for parametric evaluation.

Category of FWF	More favorable FWF	Less favorable FWF
Cement	FUETAP	PCC
Glass	ASG	BSG
Ceramic	IEB4	IEB

Abbreviations: FUETAP - formed under elevated temperature and pressure (cement)
PCC - Portland cement/concrete
ASG - aluminosilicate glass
BSG - borosilicate glass
IEB - iron-enriched basalt
IEB4 - enhanced iron-enriched basalt.

Management Site of the DOE's Nevada Test Site, the E-Area Vault Disposal Facility at the Savannah River Site, and the 200 W Area burial grounds at the Hanford Site. The sites are referred to herein as Sites A, B, C, D, and E, respectively.

Site A was studied because it is adjacent to the TSA. Site B was selected because its underlying sedimentary beds in the vadose zone are among the thickest at the INEL and would provide substantial retardation of radionuclides. Site C was picked because it has virtually no groundwater flow. Site D was studied in order to illustrate the effects of disposal of the INEL waste at a humid site. Site E was selected for study to illustrate the effects at another relatively well characterized, semiarid site.

The study of a disposal site in this PA does not imply in any way that the site will be the actual disposal location for the INEL ALLW.

Scenarios, Transport Pathways, and Exposure Routes

To assess the performance of ALLW disposal systems, two scenarios were evaluated.

The first is the undisturbed-waste scenario, i.e., the scenario in which no intrusion occurs (see Figure ES-1). After 100 yr of institutional control, members of the public were assumed to use a drinking water supply well located 100 m downgradient from the edge of the disposal unit.

The full range of transport pathways was considered for this scenario. However, based on other studies, it was concluded that (a) groundwater transport with a full range of exposure routes and (b) atmospheric transport of radon with outdoor inhalation exposure would be the dominant contributors to dose. These pathways were evaluated (see Figure ES-2).

Figure ES-1. Scenarios evaluated.

Figure ES-2. Transport pathways and exposure routes evaluated.

The groundwater transport simulations were continued until the peak radionuclide concentrations appeared at the receptor well, in some cases millions of years after closure of the disposal unit. The radon calculations covered times up to 167,000 yr, when the radon production rate peaks in this scenario.

The second scenario is the intrusion scenario, which is assumed to occur after the loss of institutional control. The scenario has four parts (see Figures ES-1 and ES-2). In the intruder-drilling scenario, an intruder drills a well directly through the source zone (the waste disposal unit). The drilling crew is exposed to the radionuclides in the exhumed waste/soil mixture from external radiation and inhalation. In the intruder-excavation scenario, waste/soil mixture is excavated during construction activities. Onsite workers are exposed by external radiation and inhalation. In the intruder-agriculture scenario, some of the waste/soil mixture is exhumed by the drilling or excavation activities just described. The mixture is spread over the ground surface, resulting in exposure by direct radiation and inhalation. In addition, radionuclides are taken up by crops and are ingested with foodstuffs of plant and animal origin. In the intruder-radon scenario, an intruder resides in a house with a basement that is constructed directly over the waste. Radon created by the radioactive decay of radionuclides in the waste is released from the FWF and diffuses through the soil and the concrete walls of the basement. The radon then circulates through the house and basement, where the indoor intruder receives a dose via the inhalation exposure route.

The waste was assumed to be buried in a large pit. The waste stack height was 22.7 ft (6.9 m), corresponding to eight 55-gal drums. The top of the uppermost drums was either 5 ft or 9 ft belowgrade (two cases), with the void being backfilled with soil. An 11-ft-thick, multilayer, abovegrade cover was emplaced over the backfill. The cover was designed in accordance with Nuclear Regulatory Commission (NRC) and Environmental Protection Agency (EPA) guidelines to help ensure its integrity for 500 yr. After that time, it was assumed conservatively not to be present. (The belowgrade backfill was assumed to be present for all times.) Thus, the initial total thickness of geologic material over the waste was either 16 ft or 20 ft, depending upon which of the two cases was being studied.

Analysis of Performance

Conceptual models were developed to describe the dominant physical and chemical processes that were assumed to occur in the various scenarios and transport pathways.

The conceptual models were implemented by a suite of numerical models. For the intruder-radon scenario, the RESRAD computer code was used to perform the assessments. For the other intruder scenarios, the Generation-II (GENII) package of computer codes was used. For the groundwater-transport pathway in the undisturbed-waste scenario, the Variably Saturated 2-Dimensional Flow and Transport code (VS2DT) was used to model the flow of water through the source zone. The Disposal Unit Source Term (DUST) code was used to model the release of radionuclides and their transport within the source zone. Flow and transport in the vadose zone (below the waste disposal pit) and in the saturated zone were evaluated using analytical solutions. Finally, exposure routes were evaluated using the Dose Integrated Over Ten Thousand Years (DITTY) code of the GENII package. For radon transport in the undisturbed-waste scenario, analytical solutions were used.

The performance-related characteristics of the six FWFs, such as leach rates, were estimated using a combination of the limited existing laboratory data and theoretical calculations. Site characteristics were estimated using a combination of the results of questionnaires completed by site representatives and data found in published reports.

For the undisturbed-waste scenario-groundwater transport pathway, one generic model was used to model radionuclide transport at the five sites, which have widely varying characteristics. Therefore, the uncertainties in the predicted doses via this pathway, in particular, are sizeable.

Results and Conclusions

About 120 runs were made, using the numerical models, to evaluate performance in the undisturbed-waste scenario and the intrusion scenarios. Because of the large volume of results, only the conclusions drawn from those results are presented here. Based on the cases studied, the conclusions are as listed below.

- Compliance with the performance objectives of DOE Order 5820.2A depends not only on the FWF and the disposal site, but also on the overall conservatism of the assumptions in the PA.
- For the undisturbed-waste scenario—atmospheric transport pathway:
 - All FWFs at all sites would be expected to comply with the performance objectives. Although no existing standards apply to outdoor releases of radon from disposed ALLW, the predicted values for radon flux, concentration, and dose are much less than the nonapplicable standards.
- For the undisturbed-waste scenario—groundwater transport pathway, compliance with the performance objective on dose:
 - At Site C, any FWF would be expected to comply.
 - At Sites A and B, nearly all FWFs would be expected to comply under nearly all assumptions. Possible exceptions under very conservative assumptions are the untreated FWF and (if no credit is taken for the chemical interactions of the radionuclides with the cement) the PCC and FUETAP FWFs.
 - At Sites D and E, only IEB4 at Site E satisfies the performance objective, and that by only a very small margin.
 - The dominant radionuclide contributing to dose is generally the Am-241 decay chain for Sites A, B, and C. For Sites D and E, the dominant contributor is the U-233 decay chain in most cases.
 - The results depend particularly on the assumed values for the leach rate of the FWF, the infiltration rate, and the K_d values at the particular site.

- For the undisturbed-waste scenario—groundwater transport pathway, compliance with the performance objective on groundwater protection:
 - All FWFs pass at Sites A, B, and C.
 - All FWFs fail at Sites D and E.

As stated previously, the use of one generic model for the groundwater pathway and the disposal unit design in addressing five sites with widely varying characteristics limits the accuracy to which the current method can predict radionuclide transport and dose via this pathway. Therefore, the conclusions should be viewed as scoping results only and subject to later confirmation by site-specific modeling.

- For the intrusion scenarios:
 - To the accuracy of these analyses, the results of the intrusion scenario were assumed to be independent of the location of waste disposal.
 - For the intruder-drilling scenario, all FWFs and burial depths studied satisfy the performance objective. (The one exception was the highly unlikely refined-inventory case in which the intruder drills into the remote-handled waste rather than into the contact-handled waste or an assumed homogeneous mixture of remote-handled and contact-handled waste.)
 - For the intruder-excavation scenario, all FWFs and burial depths studied satisfy the performance objective. (The one exception was the case involving TRU waste, not ALLW).
 - For the intruder-agricultural scenario with the 5-ft burial depth (16-ft total cover), all FWFs fail to satisfy the performance objective.
 - For the intruder-agricultural scenario with the 9-ft burial depth (20-ft total cover), the PCC, FUETAP, BSG, ASG, and untreated waste satisfy the performance objective (assuming no intrusion into the remote-handled waste). The IEB and IEB4 FWFs pass if (a) the refined inventories are used or (b) a refined particle size distribution is used to reflect the breakup of the FWF as a result of the intrusion.
 - For the intruder-radon scenario, all FWFs pass.
 - The dominant radionuclides contributing to dose are Cs-137 and Am-241 during the early period studied (e.g., 100 yr postclosure) and Am-241 and Pu-239 for the later period (500 yr postclosure). Separate runs confirmed that the times of 100 yr and 500 yr are the times of peak dose.
 - The results depend particularly on the depth of burial (belowgrade), cover longevity, and radionuclide concentration in the FWF.

- Improved immobilization of the radionuclides into a more concentrated FWF can degrade the intrusion performance by concentrating the source term, unless credit is taken for the likely larger particle sizes of the improved FWF when subjected to drilling.

Recommendations for Decisionmaking

The following observations and recommendations are offered, based on the information developed in this preliminary parametric PA:

- PA considerations should continue to be part of the waste treatment project. As the project moves forward and focuses more closely on specific sites and specific FWFs, the PA studies should become more focused. Several topics meriting further study have been identified.
- Disposal of the INEL ALLW at Site D or Site E or other sites with similar characteristics is not expected to satisfy the performance objectives, regardless of the FWF. However, the uncertainties in the current generic modeling of groundwater transport are particularly sizeable at Sites D and E.
- It is expected that disposal of the INEL ALLW at Sites A, B, or C or other sites with similar characteristics can satisfy the performance objectives.
 - Disposal will need to be at least 9 ft below grade.
 - Credit may need to be taken for the chemical interactions in the PCC and FUETAP FWFs to predict satisfactory performance in the undisturbed-waste scenario—groundwater transport pathway.
 - The improved radionuclide inventory or the refined particle size distribution may have to be used for the IEB and IEB4 FWFs to predict satisfactory performance in the intruder-agricultural scenario.
 - Special provisions are expected to be needed for disposal of the remote-handled ALLW. Disposal of such waste in conventional containers in pits is not expected to satisfy the performance objectives. Possible provisions include disposing of such waste by intermediate-depth disposal or at a deep geologic repository.

CONTENTS

ABSTRACT	iii
EXECUTIVE SUMMARY	v
ACRONYMS	xxv
1. INTRODUCTION	1-1
1.1 Private Sector Participation Initiative	1-1
1.2 Purpose, Need for, Scope, and Use of Preliminary Parametric Performance Assessment	1-3
1.3 Multiphase Nature of the Study	1-4
1.4 Requirements To Be Met in Formal Performance Assessment	1-5
1.4.1 General	1-5
1.4.2 Performance Objectives	1-5
2. THE WASTE	2-1
2.1 Radioactive Waste Management Complex	2-1
2.2 Brief Description of the Stored Alpha Low-Level Waste	2-1
2.3 Selection of Final Waste Forms for Parametric Evaluation	2-4
2.4 Radionuclide Inventory in the ALLW	2-4
2.4.1 Introduction	2-5
2.4.2 Simplified Radionuclide Inventory Used for Most of the Computer Runs	2-6
2.4.3 Refined Inventory for Nonactinides	2-11
2.4.4 Inventory with Additional Refinements for Nonactinides	2-13
2.4.5 Refined Modeling of Untreated Waste	2-15
2.5 Radionuclide Screening of Nonactinides from Detailed Groundwater Pathway Evaluation	2-20
2.5.1 Objective and Scope	2-20
2.5.2 Approach and Generic Assumptions	2-20
2.5.3 Screening Calculations for the Simplified Inventory	2-21
2.5.4 Screening Calculations for the Inventory with Additional Refinements	2-23

2.5.5	Screening Calculations for the Refined Inventory	2-25
3.	THE DISPOSAL SITES	3-1
3.1	The Sites Studied	3-1
3.2	Site A (The Subsurface Disposal Area of the Radioactive Waste Management Complex)	3-4
3.3	Site B (Site 14)	3-4
3.4	Site C (The Radioactive Waste Management Site at the Nevada Test Site)	3-5
3.5	Site D (E-Area Vaults Disposal Facility at Savannah River Site)	3-5
3.6	Site E (200 W Area Burial Grounds at Hanford Site)	3-6
3.7	Sorption Coefficients for Radionuclides for Sites A, B, and C	3-6
3.8	Sorption Coefficients for Radionuclides for Site D	3-8
3.9	Sorption Coefficients for Radionuclides for Site E	3-8
4.	SCENARIOS, TRANSPORT PATHWAYS, AND EXPOSURE ROUTES EVALUATED	4-1
4.1	Introduction	4-1
4.2	Scenarios, Transport Pathways, Exposure Routes, and Receptors	4-2
4.2.1	Undisturbed-Waste Scenario	4-2
4.2.2	Intrusion Scenario	4-4
4.3	Time Periods	4-5
5.	EVALUATION OF PERFORMANCE—INTRODUCTION	5-1
5.1	Introduction	5-1
5.2	Model of Waste Disposal Pit	5-2
6.	EVALUATION OF PERFORMANCE IN UNDISTURBED-WASTE SCENARIO—GROUNDWATER TRANSPORT PATHWAY	6-1
6.1	Introduction	6-1
6.2	Source Zone	6-1

6.2.1	Conceptual Model for Flow and Transport	6-1
6.2.2	Numerical Model for Flow and Transport	6-4
6.2.3	Assumptions and Parameter Values	6-5
6.3	Vadose Zone	6-15
6.3.1	Conceptual Model for Flow and Transport	6-15
6.3.2	Numerical Model for Flow and Transport	6-15
6.3.3	Assumptions and Parameter Values	6-16
6.4	Saturated Zone	6-17
6.4.1	Conceptual Model for Flow and Transport	6-17
6.4.2	Numerical Model for Flow and Transport	6-17
6.4.3	Assumptions and Parameter Values	6-18
6.4.4	Handling of Radioactive Decay and Ingrowth During Groundwater Transport	6-18
7.	EVALUATION OF PERFORMANCE IN UNDISTURBED-WASTE SCENARIO—EXPOSURE ROUTES FOR GROUNDWATER TRANSPORT PATHWAY	7-1
7.1	Introduction	7-1
7.2	Ingestion	7-1
7.2.1	Conceptual Model	7-1
7.2.2	Numerical Model	7-1
7.2.3	Assumptions and Parameter Values	7-2
7.2.4	Effective Dose Conversion Factors for Groundwater Pathway	7-3
8.	EVALUATION OF PERFORMANCE IN UNDISTURBED-WASTE SCENARIO—ATMOSPHERIC TRANSPORT PATHWAY (RADON TRANSPORT AND RECEPTOR EXPOSURE)	8-1
8.1	Introduction	8-1
8.2	Conceptual Model	8-1
8.3	Numerical Model	8-2
8.4	Assumptions and Parameter Values	8-4
9.	EVALUATION OF PERFORMANCE IN INTRUSION SCENARIOS	9-1
9.1	Introduction	9-1

9.2	Intruder-Drilling Scenario	9-2
9.2.1	Conceptual Model	9-2
9.2.2	Numerical Model	9-3
9.2.3	Assumptions and Parameter Values	9-3
9.3	Intruder-Excavation Scenario	9-4
9.3.1	Conceptual Model	9-4
9.3.2	Numerical Model	9-4
9.3.3	Assumptions and Parameter Values	9-5
9.4	Intruder-Agricultural Scenario	9-6
9.4.1	Conceptual Model	9-6
9.4.2	Numerical Model	9-8
9.4.3	Assumptions and Parameter Values	9-8
9.5	Intruder-Radon Scenario	9-10
9.5.1	Conceptual Model	9-10
9.5.2	Numerical Model	9-11
9.5.3	Assumptions and Parameter Values	9-11
9.5.4	Comparison of Calculational Method for Intruder-Radon Scenario against that for Radon in Undisturbed-Waste Scenario	9-12
10.	RESULTS	10-1
10.1	Introduction and Summary	10-1
10.2	Results for Undisturbed-Waste Scenario: Groundwater Pathway	10-12
10.2.1	Effects of FWF on Results	10-12
10.2.2	Effects of Disposal Site on Results	10-13
10.2.3	Effects of TRU Radionuclide Activity Concentration on Results ...	10-15
10.2.4	Effects of FWF Degradation on Results	10-16
10.2.5	Effects of Water Infiltration Rate on Results	10-18
10.2.6	Effects of Assumptions Concerning Vadose Zone Basalt on Results, Separately and Combined with Infiltration Rate	10-19
10.2.7	Effects of Contaminant Sorption in Source Zone on Results, Separately and Combined with Infiltration Rate and Vadose Zone Basalt	10-21
10.2.8	Effects of Far-Field Sorption Coefficients on Results	10-23
10.2.9	Effects of Distance to Receptor Well on Results	10-25
10.2.10	Effects of Untreated Waste on Results	10-25
10.2.11	Effects of FWF Leach Rates on Results	10-28
10.2.12	Effects of Time-Dependent Leach Rates on Results	10-30

10.2.13	Effects of Chemical Interactions within Cement Final Waste Forms on Results	10-31
10.2.14	Effects of Refined Inventory on Results	10-34
10.2.15	Effects of Inventory with Additional Refinements on Results	10-34
10.2.16	Effects of Revised Uranium and Thorium Inventories on Results ..	10-36
10.2.17	Effects of Vault-Type Disposal Facility on Results	10-38
10.2.18	Reasons for Large Doses at Sites D and E	10-39
10.2.19	Compliance with Performance Objective to Protect Groundwater Resources	10-46
10.3	Results for Undisturbed-Waste Scenario: Atmospheric Pathway	10-51
10.4	Results for Intrusion Scenarios	10-52
10.4.1	Introduction and Rationale	10-52
10.4.2	Effects of FWF on Results	10-55
10.4.3	Effects of TRU Radionuclide Activity Concentration on Results ...	10-58
10.4.4	Effects of Disposal Depth on Results	10-59
10.4.5	Effects of Untreated Waste on Results	10-59
10.4.6	Effects of Refined Inventory on Results	10-62
10.4.7	Effects of Inventory with Additional Refinements on Results	10-63
10.4.8	Effects of Exposure Duration on Results for Intruder-Drilling Scenario	10-65
10.4.9	Effects of Post-Disruption Particle Size Distribution of Final Waste Form on Results	10-66
10.4.10	Effects of Scenario Timing on Dose	10-72
10.4.11	Dominant Radionuclide Contributors to Dose	10-75
10.4.12	Results for Intruder-Radon Scenario	10-82
10.5	Qualitative Prediction of the Effects of Secondary FWFs	10-83
11.	CONCLUSIONS, AND IMPLICATIONS FOR DECISIONMAKING	11-1
11.1	Conclusions	11-1
11.1.1	Undisturbed-Waste Scenario—Atmospheric Transport Pathway	11-1
11.1.2	Undisturbed-Waste Scenario—Groundwater Transport Pathway	11-2
11.1.3	Intrusion Scenarios	11-5
11.2	Implications for Decisionmaking	11-7
11.2.1	Introduction	11-7
11.2.2	Observations and Recommendations	11-7
12.	PERSPECTIVES ON THE ASSUMPTIONS AND PARAMETER VALUES USED: HOW CONSERVATIVE?	12-1

12.1	Introduction	12-1
12.2	Comparisons against the Assumptions and Parameter Values Used in Related Assessments	12-1
12.3	Results of the Comparisons	12-2
12.4	Conclusions	12-3
13.	UNCERTAINTIES AND AREAS NEEDING FURTHER STUDY	13-1
13.1	Uncertainties	13-1
13.2	Areas Needing Further Study	13-1
13.2.1	Areas that Are Sufficiently Well Defined Programmatically to be Considered for Further Study	13-1
13.2.2	Areas for Further Study	13-5
Appendix A—Estimating the Radionuclide Inventory in INEL-Stored Alpha Low-Level Waste		A-1
Appendix B—Refined Model of Untreated Alpha-Contaminated Low-Level Waste Stored at the INEL		B-1
Appendix C—Completed Questionnaires Used to Obtain Geologic and Hydrologic Properties of Disposal Locations A, B, and C		C-1
Appendix D—Description of Preprocessor and Postprocessor Programs		D-1
Appendix E—Description of Computer Programs		E-1
Appendix F—Verification and Validation of Computer Programs		F-1
Appendix G—Final Waste Form Characteristics		G-1
Appendix H—Detailed Comparison of Assumptions and Parameter Values Used in Performance Assessments and Risk Assessments Related to the RWMC Subsurface Disposal Area		H-1

FIGURES

2-1.	Location and simplified layout of the Radioactive Waste Management Complex at the INEL.	2-2
3-1.	INEL disposal locations used in the study.	3-1

4-1.	Scenarios evaluated.	4-3
4-2.	Transport pathways and exposure routes evaluated.	4-3
4-3.	Time periods of the waste disposal facility.	4-6
5-1.	Conceptual layout of waste disposal pit.	5-3
6-1.	Simplified depiction of the conceptual model of groundwater flow and transport in the source zone, vadose zone, and saturated zone	6-2
8-1.	Simplified depiction of conceptual model for radon transport and receptor exposure. (Until 500 yr postclosure, an 11-ft abovegrade cover was also assumed to be present.)	8-2
9-1.	Exposure routes evaluated in the intruder scenarios.	9-1
9-2.	Disposal pit layout for intruder-drilling scenario, 100 yr postclosure. (At 500 yr postclosure, no credit would be taken for the abovegrade portion of the cover.)	9-5
9-3.	Disposal pit layout for intruder-excavation scenario, 500-yr postclosure with 9-ft belowgrade backfill.	9-7
9-4.	Disposal pit layout for intruder-excavation scenario, 500-yr postclosure with 5-ft belowgrade backfill.	9-7
9-5.	Disposal pit and house layout for intruder-radon scenario, >500 yr postclosure with 9-ft belowgrade backfill.	9-11
11-1.	Semiquantitative plot of compliance with performance objective for dose in undisturbed-waste scenario—groundwater transport pathway for Sites A, B, and C ...	11-3

TABLES

1-1.	Summary of radiological performance objectives applicable to the ALLW performance assessment	1-7
2-1.	Volumes used for waste stored in the TSA.	2-3
2-2.	Final waste forms selected for parametric evaluation	2-5
2-3.	Total simplified inventory, assuming 100 nCi/g TRU radionuclides.	2-8
2-4.	Comparison of simplified inventory (assuming upper limit of 100 nCi/g TRU radionuclides) against inventory developed from Apel et al. (1993) by Tozer (1993). ...	2-9

2-5.	Volumetric concentrations of each radionuclide in each final waste form, in Ci/m ³ , assuming 100 nCi/g TRU radionuclides for cement.	2-12
2-6.	Refined inventory of nonactinides for CH ALLW.	2-14
2-7.	Refined inventory of nonactinides for RH ALLW.	2-14
2-8.	Nonactinide inventory with additional refinements for CH ALLW.	2-16
2-9.	Nonactinide inventory with additional refinements for RH ALLW.	2-16
2-10.	Nonactinide inventory with additional refinements for CH plus RH ALLW.	2-17
2-11.	Refined model of the untreated ALLW.	2-19
2-12.	Screening of nonactinides from the simplified inventory before detailed evaluation of groundwater transport—Sites A, B, and C.	2-22
2-13.	Screening of nonactinides from the simplified inventory before detailed evaluation of groundwater transport—Sites A, B, and C with minimum groundwater travel time of 6.3 yr	2-23
2-14.	Screening of nonactinides from the inventory with additional refinements before detailed evaluation of groundwater transport—Site C (most favorable).	2-25
2-15.	Screening of nonactinides from the inventory with additional refinements before detailed evaluation of groundwater transport—Site D (least favorable).	2-24
2-16.	Screening of nonactinides from the simplified inventory before detailed evaluation of groundwater transport—Site D (least favorable).	2-26
3-1.	Values used for geologic and hydrologic properties of disposal locations.	3-3
3-2.	Values used for sorption coefficients at Sites A, B, and C.	3-7
3-3.	Values used for sorption coefficients at Site D (SRS).	3-8
3-4.	Values used for sorption coefficients at Site E (Hanford).	3-9
5-1.	Summary of conceptual models and numerical models selected for simulating processes in the performance assessment model	5-2
6-1.	van Genuchten parameters used in VS2DT unsaturated flow model for source zone ..	6-6
6-2.	Effective diffusion coefficients of nuclides in a cement FWF	6-10

6-3.	Comparison of calculated versus measured diffusion coefficients for a portland cement FWF (values in cm^2/sec)	6-10
6-4.	Values for ceramic FWF dissolution rates	6-12
6-5.	Values for glass FWF dissolution rates	6-13
6-6.	Actinide solubility limit statistics	6-15
7-1.	Parameter values for the DITTY code of the GENII package used for undisturbed-waste scenario	7-3
7-2.	Effective dose conversion factors for the various radionuclides used in DECAYDOS	7-4
9-1.	Parameter values for intruder-drilling scenario	9-4
9-2.	Parameter values for intruder-excavation scenario for cases involving 1-m (3-ft) penetration of the waste	9-6
9-3.	Parameter values for intruder-agricultural scenario	9-9
9-4.	Fractional leach rates used in the intruder-radon scenario.	9-12
9-5.	Parameters used in intruder-radon scenario.	9-13
9-6.	Similarities and differences in the methods for evaluating radon transport and exposure	9-13
10-1.	Performance objectives and top-level results	10-2
10-2.	Compilation of results for the undisturbed-waste scenario—groundwater transport pathway	10-3
10-3.	Compilation of results for the intrusion scenarios	10-9
10-4.	Effects of FWF on performance at Site B	10-12
10-5.	Effects of disposal site on performance	10-13
10-6.	Effects of FWF and saturated-zone groundwater velocity on performance at Site C	10-14
10-7.	Effects of TRU activity concentration on performance of IEB at Site B	10-15
10-8.	Effects of FWF degradation on performance at Site B	10-17

10-9. Effects of infiltration rate on performance of PCC at Sites A, B, D, and E	10-19
10-10. Combined effects of high infiltration rate and no credit for travel time through vadose zone basalt on performance at Site A	10-20
10-11. Effects of adsorption of radionuclides by backfill within the disposal unit	10-22
10-12. Effects of reduced K_d values on performance	10-24
10-13. Effects of distance to receptor well on performance	10-25
10-14. Effects of disposing of untreated waste	10-26
10-15. Effects of assumed release mechanism for untreated waste	10-27
10-16. Leach rates for the final waste forms	10-29
10-17. Effects of leach rates on performance	10-29
10-18. Recommended and selected values for the two-stage leach rate model, g/m^2 -day ...	10-30
10-19. Effects of time-dependent FWF leach rates on performance	10-31
10-20. Effects of chemical interactions of radionuclides with cement	10-33
10-21. Effects of inventory with additional refinements on performance	10-35
10-22. Effects of possible increases in U-233 and Th-232 inventory on performance	10-37
10-23. Effects of vault-type facility for given disposal sites	10-40
10-24. Comparison of assumptions and results of PA runs involving Sites D and E	10-41
10-25. New runs to help clarify the large differences in PA runs for INEL ALLW disposed of at Sites D and E. (All runs involve INEL ALLW. Because of the complex, focused nature of the set of runs listed here, they are not included in 10-2.)	10-45
10-26. Compliance with performance objective to protect groundwater resources	10-47
10-27. Dose from radon release	10-51
10-28. Burial depths and intrusion times evaluated for the intrusion scenarios	10-54
10-29. Cases evaluated for the intrusion scenarios.	10-56
10-30. Effects of FWF on performance in intrusion scenarios	10-57

10-31. Effects of TRU activity concentration in IEB on performance in intrusion scenarios	10-58
10-32. Effects of burial depth on performance in intrusion scenarios	10-60
10-33. Intrusion scenario results for untreated waste	10-61
10-34. Intruder scenario doses for refined inventory	10-63
10-35. Intruder scenario doses for inventory with additional refinements	10-64
10-36. Effects of exposure duration in the intruder-drilling scenario	10-66
10-37. Effects of particle size distribution on results for intruder-drilling scenario for 20-ft cover, 100 yr	10-70
10-38. Effects of particle size distribution on results for intruder-agricultural scenario for 20-ft cover, 100 yr	10-71
10-39. Time dependence of dose in intruder-drilling scenario with 16-ft soil cover	10-73
10-40. Time dependence of dose in intruder-excavation scenario with 16-ft soil cover	10-74
10-41. Time dependence of dose in intruder-agricultural scenario initiated by drilling with 16-ft soil cover	10-75
10-42. Time dependence of dose in intruder-agricultural scenario initiated by excavation with 16-ft soil cover	10-76
10-43. Time dependence of dose in intruder-agricultural scenario initiated by drilling with 20-ft soil cover	10-76
10-44. Dominant radionuclides contributing to dose for the intrusion scenarios	10-77
10-45. Results for the intruder-radon scenario	10-83
10-46. Regulatory requirements and results for intruder-radon scenario	10-84
12-1. Degree of conservatism of the most important assumptions and parameter values used in this study	12-4
13-1. Key uncertainties in this study.	13-2

ACRONYMS

ALLW	alpha low-level waste
ASG	aluminosilicate glass
BDAT	best demonstrated available technology
BSG	borosilicate glass
C&SB	Certified and Segregated Waste Storage Building
CEDE	committed effective dose equivalent
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
CFR	Code of Federal Regulations
CH	contact-handled
DITTY	Dose Integrated Over Ten Thousand Years (computer program)
DOE	(U.S.) Department of Energy
DUST	Disposal Unit Source Term (computer program)
EAVDF	E-Area Vault Disposal Facility
EDE	effective dose equivalent
EPA	(U.S.) Environmental Protection Agency
FFA/CO	Federal Facility Agreement and Consent Order
FUETAP	formed under elevated temperature and pressure
FWF	final waste form
GENII	Generation-II (computer program)
IEB	iron-enriched basalt
IEB4	enhanced iron-enriched basalt
ILTSF	Intermediate-Level Transuranic Storage Facility
INEL	Idaho National Engineering Laboratory
IWPF	Idaho Waste Processing Facility
LANL	Los Alamos National Laboratory
LDR	land disposal restriction
LLW	low-level waste
MAP	mixed activation products
MFP	mixed fission products
MPC	maximum permissible concentration

NRC	(U.S.) Nuclear Regulatory Commission
NTS	Nevada Test Site
PA	performance assessment
PCC	Portland cement/concrete
PSPI	Private Sector Participation Initiative
RCRA	Resource Conservation and Recovery Act
RFP	Rocky Flats Plant
RH	remote-handled
RWMC	Radioactive Waste Management Complex
RWMIS	Radioactive Waste Management Information System
RWMS	Radioactive Waste Management Site
SDA	Subsurface Disposal Area
SRS	Savannah River Site
SWEPP	Stored Waste Examination Pilot Plant
TRU	transuranic
TSA	Transuranic Storage Area
TSCA	Toxic Substances Control Act
UTR	untreated waste
VS2DT	Variably Saturated Two-Dimensional Flow and Transport (computer program)
WIPP	Waste Isolation Pilot Plant
ANS	American Nuclear Society

Preliminary Parametric Performance Assessment of Potential Final Waste Forms for Alpha Low-Level Waste at the Idaho National Engineering Laboratory

1. INTRODUCTION

This document presents a preliminary parametric performance assessment (PA) concerning the eventual disposal of mixed alpha low-level waste (ALLW) currently stored at the Idaho National Engineering Laboratory (INEL). The study was performed in three phases. The first two phases were performed for the Private Sector Participation Initiative (PSPI). The third phase was performed as part of the continuing effort to develop and implement methods for the ultimate disposition of the ALLW.

Section 1.1 provides background information concerning the PSPI. Section 1.2 discusses the purpose, need for, scope, and use of the results of this study. Section 1.3 discusses the multiphase nature of the study. Finally, Section 1.4 lists the performance objectives for radionuclide isolation with which a formal PA addressing the INEL ALLW must eventually demonstrate compliance.

1.1 Private Sector Participation Initiative

The Department of Energy (DOE) has a need for treatment of waste stored at the INEL. Private sector services for treating the waste are being evaluated (DOE 1995). The PSPI supports the DOE in the technical aspects of this possible procurement.

Beginning in 1970, actual or suspected transuranic (TRU) waste^a has been retrievably stored at the Radioactive Waste Management Complex (RWMC) of the INEL. The total amount

a. The current definition of transuranic (TRU) waste is as follows, as stated in DOE Order 5820.2A, "Radioactive Waste Management:"

Transuranic (TRU) waste—Without regard to source or form, waste that is contaminated with alpha-emitting transuranium radionuclides with half-lives greater than 20 yr and concentrations greater than 100 nCi/g at the time of assay.

Some of the suspected TRU waste eventually will be determined to be low-level waste (LLW):

Low-level waste—Waste that contains radioactivity and is not classified as high-level waste, transuranic waste, spent nuclear fuel or 11e(2) by-product material. . . .

Before 1984, the lower limit of transuranic radionuclide activity for defining TRU waste was 10 nCi/g, rather than the currently specified 100 nCi/g.

Much of the LLW and TRU waste stored in the Transuranic Storage Area (TSA) is mixed waste: waste containing both radioactive and hazardous chemical components as defined by the Atomic Energy Act and the Resource Conservation and Recovery Act (RCRA), respectively.

At the INEL, the following additional definition is used:

Alpha low-level waste (ALLW)—LLW that is contaminated with alpha-emitting transuranium radionuclides with half-lives greater than 20 yr and concentrations >10 and ≤ 100 nCi/g at the time of assay.

ALLW can be either mixed or not mixed. In this document, the term ALLW refers to both the mixed and the non-mixed ALLW, unless otherwise indicated.

of this stored waste is 63,729 m³ (Raivo et al. 1993). An estimated 24,413 m³, or almost 40% of the waste, is ALLW (Raivo et al. 1993).^b This waste is the subject of the PSPI and of this study.

Because most of the ALLW is considered a mixed waste, disposal of the waste may be restricted by the land disposal restrictions (LDRs) of the U.S. Environmental Protection Agency (EPA). Also, some of the ALLW may contain constituents regulated under the Toxic Substances Control Act (TSCA).

Appropriate treatment and disposal of the ALLW is being planned to ensure long-term protection of human health and the environment. The treatment must ensure that the final waste form (FWF) will satisfy the applicable disposal requirements.

The current strategy for the portion of the stored waste determined by radioassay to be TRU waste is as follows:

- Retrieve the waste
- Perform minimal processing to meet the acceptance criteria of the Waste Isolation Pilot Plant (WIPP)
- Ship it when WIPP is approved to receive such waste.

The ALLW is expected to be segregated from the TRU waste by means of the radioassay results. The ALLW is excluded from disposal at WIPP. It is also currently excluded from disposal at the RWMC Subsurface Disposal Area (SDA).

The current strategy for addressing the ALLW is as follows:

- Fund and evaluate several feasibility studies on private sector solutions for treatment of the waste
- Procure production-scale waste treatment services (use of private-sector treatment services is expected to be more rapid and cost effective)
- Select a suitable disposal location for the treated waste.

The feasibility studies have been completed and are undergoing evaluation.

The above strategies are subject to modification as the evaluations of management alternatives for the TRU waste and ALLW continue.

b. The estimates given in Raivo et al. (1993) were the best available at the time this study began. Subsequent evaluations have resulted in slightly different estimates. The differences are not expected to have a substantial effect on the results of the current study.

1.2 Purpose, Need for, Scope, and Use of Preliminary Parametric Performance Assessment

The purpose of this preliminary parametric PA was to provide preliminary evaluations of the performance of the treated ALLW—in various FWFs, in various disposal locations, and under various assumptions.

Not only must treatment of the ALLW satisfy EPA requirements, but the waste disposal system must also satisfy performance objectives in DOE Order 5820.2A. The performance objectives are stated in terms of the projected radiological doses to members of the public and to inadvertent intruders after the loss of institutional control. The performance objectives also require the protection of groundwater resources, consistent with regulatory requirements. See Section 1.4 for details. The disposal system consists of the FWF, the packaging materials, any engineered barriers, and the geological environment of the disposal site. The evaluation of the projected performance is called a PA. Thus, eventually a formal PA must be prepared addressing the performance of the treated INEL ALLW in the chosen disposal setting.

It is too early in the PSPI project to prepare a formal PA. However, this study fulfills a related need: early, preliminary projections of whether a given FWF at a given disposal site might be expected to satisfy the performance objectives, and if so, by how much.

Implementing the PSPI or any other large-scale treatment operation involves the commitment of large amounts of funding. The treatment method(s) selected must provide the appropriate degree of immobilization of the radionuclides. If the treatment provides insufficient immobilization, the disposal system (including the FWF) will not comply with the performance objectives. If the treatment provides an unnecessary degree of immobilization, more funding may be expended than is necessary. Thus, the selection of a FWF is a crucial decision. This preliminary parametric PA provides vital early indications of the extent of immobilization that may be necessary. This study also examines the interactions among the FWF, the waste disposal location, and the other parameters that affect the performance of the waste disposal system.

This study is termed a preliminary, parametric PA:

- It is preliminary in that it is not the PA that will be prepared for formal approval in accordance with DOE Order 5820.2A. At present, only a preliminary PA can be prepared because, among other reasons, the FWF and disposal location have not been selected.
- It is parametric in that several parameters were varied to observe their effect on the performance of the disposal system. Included were the FWF, the disposal location, the depth of burial, the degree of conservatism of the assumed values of geohydrologic parameters, etc. The study did not examine one FWF and one disposal location in detail. A parametric approach was selected because, at this early time in the PSPI program, a wide range of options required evaluation.

This study has several potential uses:

- It will help guide program managers in determining the optimum degree of immobilization for the ALLW to satisfy the performance objectives while minimizing cost
- The results indicate the margin of compliance with the performance objectives and how it depends on various parameters
- It identifies key parameters that have a major effect on the performance and for which better data are needed to support decisions.

Two related evaluations of waste disposal performance at the SDA provided much of the input information concerning disposal performance in the SDA. Maheras et al. (1994) is the formal PA that supports ongoing disposal of LLW at the RWMC SDA. It addresses LLW buried or projected to be buried from 1984 through a potential closure date of 2020. It was prepared pursuant to DOE Order 5820.2A. Various drafts of the formal PA, extending back as far as 1990 (Case et al. 1990), were consulted.

Loehr et al. (1994) is a preliminary scoping risk assessment. It was prepared pursuant to the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) and an EPA/State of Idaho/DOE agreement [the Federal Facility Agreement and Consent Order (FFA/CO)] concerning possible remediation of part of the SDA. It addresses all waste buried belowgrade from 1952 through 1983. Follow-on risk assessments will extend the 1983 cutoff through 1994.

1.3 Multiphase Nature of the Study

This study was performed in three phases, all of which are reported herein. In the first phase, PA computer runs were performed using an initial set of assumptions and parameter values. In the second and third phases, the results obtained with the initial assumptions and parameter values were evaluated to identify topics needing further study. Additional runs were then made using refined assumptions or expanded ranges for some parameter values. The complete set of initial runs was not repeated using the revised assumptions or parameter values.

The multiphase nature of the study added complexity that was manifest in several ways. For example, in some cases, determining the effects of changing only one parameter at a time necessitated using, as a reference for comparison, early combinations of parameter values that had been determined later to be less than optimum. As another example, not every interesting combination of parameter values could be studied because of the large number of such combinations.

Some aspects of this report reflect the fact that the study was performed in successive phases and was an iterative process. The text reflects some revisiting of early results that was necessary because of knowledge gained in later phases. Also, the large number of runs with various

combinations of values required elaborate tables to track the input values used for each run so the individual results could be interpreted in the proper context.

In spite of the added complexity, the study produced a wide range of useful information on the intricate dependencies of the many parameters that affect a PA.

1.4 Requirements To Be Met in Formal Performance Assessment

1.4.1 General

Chapter III of DOE Order 5820.2A deals with the management of LLW, under which the INEL ALLW is categorized. Paragraph 3b(1) of Chapter III requires each DOE field organization with a disposal site to "prepare and maintain a site-specific radiological performance assessment for the disposal of waste for the purpose of demonstrating compliance with the performance objectives stated in paragraph 3a." The formal review process for PAs is also described in the order.

1.4.2 Performance Objectives

Paragraph 3a of Chapter III lists the performance objectives for DOE LLW not disposed of before issue of the Order on September 26, 1988. The objectives are as follows:

1. Protect public health and safety in accordance with standards specified in applicable EH orders and other DOE orders.
2. Ensure that external exposure to the waste and concentrations of radioactive material which may be released into surface water, groundwater, soil, plants and animals results in an effective dose equivalent (EDE)^c that does not exceed 25 mrem/yr to any member of the public. Releases to the atmosphere shall meet the requirements of 40 Code of Federal Regulations (CFR) 61 (EPA 1992a). Reasonable effort should be made to maintain releases of radioactivity in effluents to the general environment as low as is reasonably achievable.
3. Ensure that the committed effective dose equivalents (CEDE) received by individuals who inadvertently may intrude into the facility after the loss of active institutional control (100 yr) will not exceed 100 mrem/yr for continuous exposure or 500 mrem for a single acute exposure.
4. Protect groundwater resources, consistent with Federal, State, and local requirements.

Item 2 above cites 40 CFR 61, "National Emission Standards for Hazardous Air Pollutants (Clean Air Act)," wherein the following performance objective is given:

c. Hereafter, the term dose or, where appropriate, peak annual dose is generally used in place of the effective dose equivalent or peak annual EDE.

- The airborne effluent pathway shall not result in any member of the public receiving, in a year, a dose equivalent >10 mrem EDE.

Item 4 above, related to the protection of groundwater resources, was interpreted here to mean that concentrations of radionuclides should not exceed the standards for public drinking water supplies as established by the EPA. The National Primary Drinking Water Regulations in 40 CFR 141 (EPA 1992b) state that all community drinking water systems that have at least 15 connections or serve at least 25 people must meet the following requirements:

- Beta and photon (gamma) emitters, ≤ 4 mrem/yr
- Total alpha emitters, ≤ 15 pCi/L (including Ra-226, but excluding uranium and radon)
- Combined Ra-226 and Ra-228, ≤ 5 pCi/L.

The standard used for uranium in water is that given in a proposed rule-making (EPA 1991) to 40 CFR 141:

- Uranium, ≤ 30 pCi/L.

The following criterion (EPA 1987) was used for radon in air:

- Indoor radon concentration, ≤ 4 pCi/L.

In addition, State regulations may apply to the ALLW disposal system and must be at least as stringent as the Federal regulations. In the case of Idaho, the requirements are the same as the Federal requirements. No search was made of the requirements of other states in which the INEL ALLW could conceivably be disposed.

For convenience, the performance objectives are summarized in Table 1-1.

Proposed revisions to DOE Order 5820.2A have been drafted. The performance objectives that will appear in the final revision of the order are not known at this time. It seems prudent to use the current performance objectives in the interim.

Although this preliminary parametric PA was not prepared for the formal approval of a waste disposal system, the same performance objectives were used as will be used for the formal PA. The intent is that this study facilitate a smooth transition later to the formal PA.

Table 1-1. Summary of radiological performance objectives applicable to the ALLW performance assessment.

Regulation	Exposure group	Performance objectives limit	Compliance point	Relevant compliance period
<u>DOE Orders</u>				
5820.2A, Ch. III	Public	25 mrem/yr EDE (all pathways but air)	Point 100 m downgradient from disposal unit boundary	During post-institutional control
	Inadvertent intruder	100 mrem/yr EDE (chronic exposure)	Disposal unit	During post-institutional control
		500 mrem EDE (acute exposure)		
<u>EPA</u>				
40 CFR 61, subpart H	Public	10 mrem/yr EDE (air emissions)	Point of maximum annual air concentration in an unrestricted area where the public resides or abides	All periods
40 CFR 141	Public	4 mrem/yr from beta emitters and photon emitters	Any community drinking water system at point 100 m downgradient from disposal unit boundary	All periods
		15 pCi/L total alpha emitters (including Ra-226, excluding U and Rn)		
		5 pCi/L combined Ra-226 and Ra-228		
		30 pCi/L U		
EPA (1987)	Inadvertent intruder	4 pCi/L Rn concentration in indoor air	Hypothetical residence overlying disposal unit	During post-institutional control

References for Section 1

- Case, M. J., S. J. Maheras, M. A. McKenzie-Carter, M. E. Sussman, and P. Voilleque, 1990, *Radioactive Waste Management Complex Performance Assessment (Draft)* EG&G Idaho, Inc., EGG-WM-8773, June.
- DOE, 1995, *Evaluation of Feasibility Studies for Private Sector Treatment of Alpha and TRU Mixed Wastes*, U.S. Department of Energy, Idaho Operations Office, DOE/ID-10512, May.
- EPA, 1987, *Interim Protocols for Screening and Follow-up Radon and Radon Decay Product Measurements*, EPA 520/1-86-014-1, U.S. Environmental Protection Agency, Office of Radiation Programs.
- EPA, 1991, "National Primary Drinking Water Regulations: Radionuclides," Federal Register, Vol. 56, No. 138 (56 FR 33050), U.S. Environmental Protection Agency.
- EPA, 1992a, "National Emission Standards for Hazardous Air Pollutants (Clean Air Act)," 40 CFR 61, U.S. Environmental Protection Agency.
- EPA, 1992b, "National Primary Drinking Water Regulations," 40 CFR 141, U.S. Environmental Protection Agency.
- Loehr, C. A., B. H. Becker, D. E. Burns, R. M. Huntley, S. M. Rood, P. Sinton, and T. H. Smith, 1994, *Preliminary Scoping Risk Assessment for Waste Pits, Trenches, and Soil Vaults at the Subsurface Disposal Area, Idaho National Engineering Laboratory*, EG&G Idaho, Inc., EGG-WM-11181, May.
- Maheras, S. J., A. S. Rood, S. O. Magnuson, M. E. Sussman, and R. N. Bhatt, 1994, *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment*, EG&G Idaho, Inc., EGG-WM-8773, May.
- Raivo, B. D., T. L. Eddy, and G. L. Anderson, 1993, *Statement of Work: Commercial Transport and Treatment of Alpha-Contaminated Mixed Low-level Radioactive Waste (draft)*, Attachment 1 to *Solid Alpha-Contaminated Low-level Radioactive Waste Transport/Treatment Services Request for Proposal*, SP-SOW-01, June.

2. THE WASTE

This section first describes the RWMC, where the subject waste is located. It discusses where, when, and how the waste was generated, and how it is currently stored. Most of the section is devoted to explaining how the radionuclide inventory in the waste was estimated before and after treatment. Finally, this section discusses radionuclide screening to simplify the evaluation of radionuclide transport via the groundwater pathway.

2.1 Radioactive Waste Management Complex^a

The RWMC is located in the southwest portion of the INEL, which is located in southeastern Idaho. The RWMC consists of the 97-acre SDA, the 58-acre TSA, and the Administrative Area. See Figure 2-1.

Since 1952, solid low-level waste has been disposed of in pits, trenches, and soil vaults in the SDA. From 1954 through 1970, solid TRU waste also was disposed of in the SDA. Although the remaining disposal space is somewhat limited, the SDA is one possible location for disposal of the ALLW after any necessary treatment.

The TSA opened in 1970 for the storage of TRU waste and suspect TRU waste. It consists primarily of the TSA pads and the Intermediate-Level Transuranic Storage Facility (ILTSF), which was added in 1976. The ALLW is stored at the TSA. The manner of storage is discussed in Section 2.2.

In the 1970s, two programs demonstrated experimental retrieval of part of the waste buried in the SDA. The volume of waste retrieved was 4,346 m³. Most of the retrieved waste was placed into storage in the TSA.

The Stored Waste Examination Pilot Plant (SWEPP) is also located at the TSA. The mission of SWEPP is to certify, by nondestructive means, containers of TRU waste to meet the waste acceptance criteria of the WIPP repository. Waste processed through SWEPP is placed in the Certified and Segregated Waste Storage Building (C&SB). SWEPP includes radiological assay equipment to measure the TRU content of waste containers, thereby allowing them to be designated as either TRU waste or ALLW.

2.2 Brief Description of the Stored Alpha Low-Level Waste

Storage of newly received TRU waste and suspect TRU waste at the TSA began in 1970.^b About 80% of the waste was generated at the Rocky Flats Plant, as residue from the production of nuclear weapons. The remainder was generated at several other DOE sites, including the

a. Most of the discussion in Section 2.1 is taken from EG&G Idaho 1985.

b. As stated in Section 2.1, some waste was retrieved from burial in the SDA and placed on the TSA. Most of that waste had been disposed of in 1970, although a small amount had been disposed of from 1955 through 1963.

ML5107

Figure 2-1. Location and simplified layout of the Radioactive Waste Management Complex at the INEL.

INEL, as a result of various operations and research programs. Most of the waste was generated from 1970 through 1989. Current receipt rates are greatly reduced from those during that period.

The waste consists of a wide variety of solid materials (e.g., sludges, paper, rags, metals, wood, soil) contaminated with small amounts of TRU radionuclides. Some of the waste, particularly that with higher radiation levels, also contains fission products and activated metals.

The contact-handled (CH) waste, defined as that with dose rates ≤ 200 mrem/hr at the container surface, is contained in steel drums, plywood boxes, and steel bins. Most of the drums have rigid polyethylene liners, and most of the boxes are reinforced with plastic-impregnated fiberglass.

The CH waste was placed on two aboveground asphalt storage pads in the TSA. See Figure 2-1. The stacked waste containers were then covered with plywood, nylon-reinforced polyvinyl sheeting, and soil.

The remote-handled (RH) waste, defined as that with dose rates > 200 mrem/hr at the container surface, is contained primarily in steel drums or inserts. Because of the relatively high radiation levels, it is placed in belowgrade, steel-and-concrete vaults in the ILTSF. See Figure 2-1.

The total volume of stored waste in the TSA is well defined: $63,729 \text{ m}^3$ (Raivo et al. 1993). See Table 2-1.

Estimating which of the stored waste is actually TRU waste and which is not TRU waste (i.e., ALLW) is less certain. The shipping records for the waste are incomplete, particularly in radionuclide information. Furthermore, during much of the time when the waste was generated,

Table 2-1. Volumes used for waste stored in the TSA.

Waste type	Volume (m^3)
ALLW	
Mixed	21,623
Nonmixed	2,790
Total	24,413
TRU waste	
Mixed	38,195
Nonmixed	1,121
Total	39,316
Grand total	63,729

the technology for determining radionuclide inventories was not as advanced as it is today. The determination of TRU content will be made by assaying the containers in SWEPP when they are retrieved for certification or treatment. The containers of treated waste will be reassayed because the effects of treatment on the activity distribution cannot be projected with confidence. Thus, only estimates of the breakdown into TRU waste vs. ALLW can be made at present. The estimates are based on the (incomplete) information in the shipping records and a limited amount of assay data from SWEPP. Table 2-1 shows the division of waste used here (Raivo et al. 1993).

The division into mixed waste and nonmixed waste is also uncertain. (It does not have a direct effect on this study). The values from Raivo et al. 1993 are used here.

Thus, as shown in Table 2-1, the total volume assumed here to be ALLW, whether mixed or not, is 24,413 m³.

2.3 Selection of Final Waste Forms for Parametric Evaluation

The FWFs selected for parametric evaluation are introduced here, along with the reasons for their selection. Radionuclide concentrations for these FWFs are presented in Section 2.4. Assumed values for the physical and chemical characteristics of the FWFs are discussed in Sections 6.2.3.3 through 6.2.3.6.

Most of the ALLW is expected to be classified as mixed waste per RCRA. Disposal of the waste may also be restricted under the LDRs due to the presence of constituents such as lead, mercury, and organics such as carbon tetrachloride. Waste containing LDR constituents must be treated using the best demonstrated available technology (BDAT) before land disposal.

The BDAT for waste contaminated with LDR organics is thermal treatment followed by solidification of the resulting ash. Options for solidification of the ash to produce the FWF include cement, glass, and ceramic waste forms. The BDAT for waste contaminated with volatile metals that are not compatible with thermal treatment, such as lead and mercury, is solidification. Options for solidification of this waste include cementation.

Glass, cement, and (non-porous) ceramic are thus considered logical candidates for the FWF of the ALLW. To evaluate a wide range of potential FWFs, two FWFs from each of these three categories were evaluated. Of each pair, one was selected so as to represent the more favorable contaminant-immobilization characteristics of that category of waste form. The other one was selected to represent the less favorable characteristics. Table 2-2 lists the six FWFs selected for parametric evaluation.

In addition, a limited number of evaluations were performed with untreated waste (UTR) as a benchmark for comparison. Because the preimmobilization treatment methods are not known, the waste evaluated was pretreatment, not preimmobilization.

Table 2-2. Final waste forms selected for parametric evaluation.

Category of FWF	More favorable FWF	Less favorable FWF
Cement	FUETAP	PCC
Glass	ASG	BSG
Ceramic	IEB4	IEB

Abbreviations: FUETAP—formed under elevated temperature and pressure (cement)
PCC—Portland cement/concrete
ASG—aluminosilicate glass
BSG—borosilicate glass
IEB—iron-enriched basalt
IEB4—enhanced iron-enriched basalt.

2.4 Radionuclide Inventory in the ALLW

2.4.1 Introduction

The methods used to develop the radionuclide inventories for this study are described in detail in Appendix A.

As stated in Section 2.2, available information on the radionuclide inventory in the stored waste is incomplete. The available information does not allow the radionuclide inventory of the ALLW to be calculated with certainty. However, approximate inventories were developed for this study by (a) using the radionuclide information in the shipping records and in the Radioactive Waste Management Information System (RWMIS) database of those records, and (b) making some assumptions, performing analyses, and obtaining inventory information from sources other than RWMIS.

Section 1.3 stated that the present parametric study was conducted in three phases. Broad assumptions were made for the first phase. Based on the results of the first phase, important parameters meriting additional study were identified. For the second and third phases, either refined assumptions or a range of assumptions was made for several of those parameters.

The waste inventory assumptions were of this type. Additional inventory information was developed, both internal and external to this study, as the study progressed. Therefore, the inventory was refined for some of the evaluations in the second and third phases.

Three successive approximations of the radionuclide inventory were developed and used for PA runs involving treated ALLW. Each approximation was more detailed and is expected to be more accurate than its predecessor.

The first version of the inventory is called the "simplified inventory." It was developed by means of the simplified approach described briefly in Section 2.4.2 and in detail in Sections A-1 through A-8 of Appendix A. It was used for all computer runs in the first phase of the study and for most of the runs in the second and third phases. Unless otherwise stated, this was the inventory used.

For the second phase, a "refined inventory," described briefly in Section 2.4.3 and in detail in Section A-9 of Appendix A, was developed and used for a limited number of runs. Those runs are so identified in the presentation of results.

For the third phase, an "inventory with additional refinements," described briefly in Section 2.4.4 and in detail in Section A-10 of Appendix A, was developed and used for a limited number of runs. Those runs are so identified in the presentation of results.

In the second phase, a refined model of the untreated waste, described briefly in Section 2.4.5 and in detail in Appendix B, was developed and used in a limited number of computer runs. Those runs are so designated in the presentation of results.

2.4.2 Simplified Radionuclide Inventory Used for Most of the Computer Runs

2.4.2.1 General. The method described below in Sections 2.4.2.2 through 2.4.2.6 was used in developing the simplified inventory for most of the computer runs.

In addition to many assumptions discussed later, three broad, simplifying assumptions were made:

- In addressing each of the six FWFs (one at a time), all waste is in the same FWF
- Within a given FWF, the radionuclide concentrations are homogeneous throughout the posttreatment waste
- All radionuclides that enter treatment with the waste stay with the treated waste.

Two groups of radionuclides are present in the waste; they were evaluated separately. The first group consists of the radionuclides with atomic number ≥ 89 and their progeny. This group of radionuclides is referred to here as the actinides. The second group consists of all other radionuclides and is referred to as the nonactinides.

For both actinides and nonactinides, the starting point for the evaluation was the shipping record-based inventory for all waste emplaced in the TSA. This pretreatment inventory is found in the RWMIS database and was decayed to 1993 by Atwood and Schlafman (1993).

Because the inventory of radionuclides in the ALLW has significant uncertainties, upper-limit and lower-limit inventories of actinides were developed to bound the actual inventory. Nearly all of the runs were made with the upper-limit inventory. For the nonactinides, the inventory is a best estimate.

2.4.2.2 Actinides. A simplified, upper-limit inventory of the actinides in the ALLW was developed by

- Assuming that their respective activities in the ALLW have the same ratio to one another as in the TSA waste as a whole
- Scaling the posttreatment inventory of actinides to an upper-limit value of 100 nCi of TRU radionuclides^c per gram of waste, the maximum possible concentration for ALLW; this step is discussed further in Section 2.4.2.6
- Screening out radionuclides with very small activities.

This inventory is conservative in that it reflects the maximum possible TRU radionuclide concentration for ALLW.

The minimum possible concentration of 10 nCi/g of TRU radionuclides in the ALLW was used to create a lower-limit inventory of actinides. Finally, a case was analyzed based on an assumption of 1,000 nCi/g of TRU radionuclides. This case actually represents TRU waste, not ALLW.

2.4.2.3 Nonactinides. A simplified inventory of nonactinides in the ALLW was developed by

- Assuming all of the nonactinides at the TSA are located in the (untreated) RH TRU waste and RH ALLW
- Assuming that the activity fraction of the nonactinide inventory that is in the RH ALLW is the same as the volume fraction of RH waste ($50\% \pm 20\%$) that is ALLW; that is, the concentration of nonactinide activity was assumed to be independent of the concentration of TRU activity
- Spreading the nonactinide activity evenly over all of the treated ALLW (RH and CH)
- Screening out radionuclides with very low activities [the screening was confirmed using the method of Seitz and Kocher (1993) for the groundwater pathway and using computer runs for the intrusion scenarios]
- Assigning specific radionuclides for generic entries such as mixed fission products.

A best estimate of the nonactinide inventory was developed because there was no basis for estimating upper-limit and lower-limit inventories.

c. Hereafter, TRU radionuclide generally refers to those radionuclides cited in DOE Order 5820.2A for the definition of TRU waste, i.e., those that are alpha-emitters and have half-lives greater than 20 yr.

2.4.2.4 Total Inventory. Table 2-3 lists the total simplified inventory (as of 1993) that was used for most of the computer runs. Radioactive decay and ingrowth after 1993 was handled in the individual schemes for transport pathway modeling and exposure route modeling.

The TRU activity is the upper limit of 100 nCi/g for ALLW. (That is, if the total TRU activity of 2,990 Ci is divided by the total mass of the waste after treatment, the result is 100 nCi/g.)

For computer runs with 10 nCi/g or 1,000 nCi/g of TRU, the activities of all actinides were, respectively, divided by 10 or multiplied by 10.

2.4.2.5 Comparison Against the Inventory Developed in Other PSPI Studies. While this study was under way, other PSPI activities included detailed evaluations of the known shipping information and SWEPP assay data on untreated waste expected to be classified as ALLW (Apel et al. 1993, Apel et al. 1994). The simplified inventory in Table 2-3 was used in this study because (a) the studies by Apel et al. were not available when the PA work started and (b) those evaluations did not identify any nonactinides in the ALLW.

Table 2-3. Total simplified inventory, assuming 100 nCi/g TRU radionuclides.

Radionuclide	Inventory in 1993 (Ci)	TRU radionuclide inventory (Ci)
Ba-137m	1,125	—
Co-60	47	—
Cs-137	1,125	—
Sr-90	1,013	—
Y-90	1,013	—
Am-241	1,340	1,340
Cm-244	8	—
Pu-238	906	906
Pu-239	598	598
Pu-240	147	147
Pu-241	2,401	—
U-233	13	—
U-238	0.0017	—
Total	9,736	2,990

Note: Totals may not match exactly because of rounding.

The inventory studies by Apel et al. addressed different information about the ALLW (e.g., characterizing untreated waste rather than treated waste). Nevertheless, comparing the current simplified inventory against that developed in the Apel studies was useful as a rough check (Smith 1993, Tozer 1993).

Table 2-4 compares the two inventories. The simplified inventory has a higher total activity and higher entries for almost all radionuclides than does the inventory compiled from Apel et al. (1993) in Tozer (1993).

The simplified inventory lists 5,413 Ci of actinides if 100 nCi/g TRU (the upper limit for ALLW) is assumed. If 10 nCi/g (the lower limit for ALLW) is assumed, the result is 541.3 Ci of actinides. These two limiting cases bound the 1,898 Ci of actinides in the inventory derived from Apel et al. (1993).

Table 2-4. Comparison of simplified inventory (assuming upper limit of 100 nCi/g TRU radionuclides) against inventory developed from Apel et al. (1993) by Tozer (1993).

Radionuclide	Simplified inventory (Ci)	Inventory from Tozer 1993 (Ci)
Ba-137m	1,125	—
Co-60	47	—
Cs-137	1,125	—
Sr-90	1,013	—
Y-90	1,013	—
Am-241	1,340	8.86
Cm-244	8	—
Pu-236	—	4E-06
Pu-238	906	126
Pu-239	598	227
Pu-240	147	49.7
Pu-241	2,401	1,330
Pu-242	—	0.06
Th-232	—	0.475
U-233	13	155
U-235	—	0.013
U-238	0.0017	0.42
Total	9,736	1,898
Total actinides	5,413	1,898

Note: Totals may not match exactly because of rounding.

As stated previously, the inventory in Apel et al. (1993) does not show any nonactinides.

With one exception, the only radionuclides for which the Apel inventory gives higher values than does the simplified inventory are all <1 Ci in activity.

The one exception is U-233. The simplified inventory lists 13 Ci, whereas Tozer's compilation from Apel et al. (1993) gives 155 Ci. The more recent Apel et al. (1994) lists approximately 865 Ci of U-233. In addition, Apel et al. (1994) lists approximately 2.5 Ci of Th-232, an increase over the 0.475 Ci in Apel et al. (1993). Because of questions about the effects of the increased activities of U-233 and Th-232, special computer runs for the groundwater pathway were made using the most recent (highest) values of U-233 and Th-232 from Apel et al. (1994). The results, discussed in Section 10, show that, for the one site studied (the SDA), the total doses increase by only about 0.1% at the most, if the highest values of U-233 and Th-232 activity are used. Thus, the larger activity of U-233 and Th-232 in Apel et al. (1994) did not have a substantial effect on the dose for that site.

A check was also made on the importance of the 0.42 Ci of U-238 in Apel et al. (1993). Although the activity is <1 Ci, the half-life of U-238 is extremely long at 4.47 billion years. A groundwater run was made for the U-238 assuming a Portland cement/concrete (PCC) FWF disposed of at the SDA. Although the U-238 produces a substantial secondary peak dose at a later time, the overall maximum dose was not increased by the larger activity of the U-238.

In conclusion, the rough comparison against the Apel inventories confirmed that the simplified inventory generally provides reasonable, bounding values for the radionuclide activities in the treated ALLW.

2.4.2.6 Treated Waste.

General and Baseline

As Section 2.4.2.2 indicated, the scaling step for actinides is discussed in more detail here.

One general assumption was that all radionuclides that enter treatment with the waste stay with the treated waste. Therefore, the radionuclide inventory, in Ci, derived for untreated waste applies also to treated waste.

The upper-limit actinide inventory was developed for the situation in which all containers of treated waste have 100 nCi/g of TRU radionuclides. In this situation, the inventory depends on the total mass of the treated waste. (The mass of the empty containers is not considered in this analysis.) To complicate things, the posttreatment mass (a) differs from the pretreatment mass and (b) depends on the treatment method.

The estimated pretreatment mass is based on a waste volume of 24,413 m³ for the ALLW (Table 2-1) and an average waste density of 652 kg/m³ from evaluations conducted in the INEL Idaho Waste Processing Facility (IWPF) Project. The result is an estimated pretreatment mass of 1.59E+10 g.

For simplicity in examining the effects on activity concentration, waste treatment can be considered to consist of two types of operations. The first type includes those that reduce the mass of the waste. For example, incineration reduces the mass of the combustible portion of the waste. However, the estimated fraction of the ALLW that is combustible is a maximum of only 25-30% (by mass), per the IWPF Project. Suppose that 25% of the waste mass were essentially eliminated (reduced to a negligible mass of ash), as a result of incineration, offgassing, etc. Therefore, each pretreatment unit of mass would be reduced to approximately 0.75 unit mass.

The second type of treatment covers operations that increase the mass of the waste. Immobilization increases the mass, by the addition of immobilization agents. Waste loadings for the FWFs (defined as mass of waste/total mass of FWF) considered range from 30 to 60%.

It was calculated that, for the FWFs considered, the midpoint of the extreme values of the total posttreatment mass of the ALLW corresponds to a 40% waste loading. (Posttreatment mass is proportional to the inverse of the waste loading.) Because it was the midpoint, this value was used as a baseline for calculational purposes. (*This midpoint loading happens to correspond to that for cement, both PCC and FUETAP. The effects for other FWFs are discussed later in this section.*) Thus, in this case, each unit mass of postincineration ALLW that undergoes (cement) immobilization increases to about 2.5 unit mass of FWF.

The cumulative effect of the two types of processes is an increase in mass from 1 unit mass of waste before treatment to approximately $0.75 \times 2.5 = 1.88$ unit mass of FWF after treatment. Therefore, the posttreatment mass is approximately $1.88 \times 1.59 \text{ E}+10 = 2.99 \text{ E}+10 \text{ g}$.

At the upper-limit concentration of 100 nCi/g, the $2.99 \text{ E}+10 \text{ g}$ has a total TRU radionuclide activity of $100 \text{ nCi/g} \times 2.99 \text{ E}+10 \text{ g} \times \text{E}-9 \text{ Ci/nCi} = 2,990 \text{ Ci}$. Therefore, the individual actinide activities were scaled to produce a total activity of 2,990 Ci of TRU radionuclides. Complete details are given in Appendix A.

Specific to Each FWF

Table 2-5 gives the volumetric concentration of activity for each radionuclide in each FWF. This information is needed because the amount of radioactivity released in the intrusion scenarios is proportional to the volumetric concentration of radioactivity. The table entries were developed by dividing the activities given in Table 2-3 by the estimated total volume of each FWF. Estimation of the posttreatment volumes is described in Section A-8 of Appendix A.

For computer runs in which the mass concentration of TRU radionuclides was 10 or 1,000 nCi/g, the values in Table 2-5 were changed as follows. For all actinides, the radioactivity and the volumetric concentration of radioactivity were decreased or increased, respectively, by a factor of 10. For all nonactinides, the values were not changed.

2.4.3 Refined Inventory for Nonactinides

The computer runs in the first phase of the study identified that the assumptions concerning the nonactinides in the inventory had a considerable effect on the calculated doses for some intrusion scenarios. Therefore, a refined inventory of the nonactinides was developed during the

Table 2-5. Volumetric concentrations of each radionuclide in each final waste form, in Ci/m³, assuming 100 nCi/g TRU radionuclides for cement.

Radionuclide	More favorable and less favorable cement (FUETAP and PCC)	More favorable glass (ASG)	Less favorable glass (BSG)	More favorable ceramic (IEB4)	Less favorable ceramic (IEB)
Ba-137m	7.34E-02	1.50E-01	7.49E-02	1.98E-01	1.70E-01
Co-60	3.07E-03	6.26E-03	3.13E-03	8.27E-03	7.09E-03
Cs-137	7.34E-02	1.50E-01	7.49E-02	1.98E-01	1.70E-01
Sr-90	6.61E-02	1.35E-01	6.74E-02	1.78E-01	1.53E-01
Y-90	6.61E-02	1.35E-01	6.74E-02	1.78E-01	1.53E-01
Am-241	8.74E-02	1.78E-01	8.92E-02	2.36E-01	2.02E-01
Cm-244	5.22E-04	1.07E-03	5.33E-04	1.41E-03	1.21E-03
Pu-238	5.91E-02	1.21E-01	6.03E-02	1.59E-01	1.37E-01
Pu-239	3.90E-02	7.96E-02	3.98E-02	1.05E-01	9.02E-02
Pu-240	9.59E-03	1.96E-02	9.79E-03	2.59E-02	2.22E-02
Pu-241	1.57E-01	3.20E-01	1.60E-01	4.22E-01	3.62E-01
U-233	8.48E-04	1.73E-03	8.66E-04	2.29E-03	1.96E-03
U-238	1.11E-07	2.26E-07	1.13E-07	2.99E-07	2.56E-07
Total	6.35E-01	1.30E+00	6.48E-01	1.71E+00	1.47E+00

Note: Totals may not match exactly because of rounding.

second phase of the study, and a few computer runs were made using this refined inventory. The inventory of actinides was not changed from that in Table 2-3. The details of developing the refined inventory are given in Section A-9 of Appendix A. An overview is given below.

In developing the simplified inventory, conservative assumptions were made concerning nonactinides in the treated ALLW. One key assumption was spreading the nonactinide activity evenly over all of the treated ALLW. The simplified approach tends to overestimate the concentrations of nonactinides in the CH ALLW and underestimate the concentration of nonactinides in the RH ALLW. For the refined inventory, this assumption was replaced by a more realistic assumption. The CH ALLW was assumed to be treated separately from the RH ALLW, and the nonactinide inventories were estimated separately. It was no longer assumed that all of the nonactinides are in the RH waste. Most of the other assumptions used to develop the refined nonactinide inventory were the same as those for the simplified nonactinide inventory discussed in Section 2.4.2.3.

It was assumed again that the distribution of nonactinides is independent of the distribution of TRU radionuclides. Thus, since 38% of the volume of the CH waste is CH ALLW, 38% of each nonactinide in the CH waste was assumed to be in the CH ALLW. The same assumption was made for the RH waste, although the fraction is 41%. (The refined value differs somewhat from the earlier estimate of $50\% \pm 20\%$ used in the simplified model.)

Separate RWMIS printouts (covering the combined ALLW and TRU waste) were produced for the CH waste and for the RH waste. The RWMIS inventory of each nonactinide in the waste was multiplied by 0.38 for the CH waste or 0.41 for the RH waste.

The refined inventory of nonactinides is given in Table 2-6 for the CH portion of the ALLW and in Table 2-7 for the RH portion.

Comparison of Table 2-6 with Table 2-3 shows a large reduction in the nonactinide inventory in the CH ALLW. (The reduction for Co-60 is small and is the only exception.) The list of radionuclides that survived the screening is unchanged. The volume of the CH ALLW is virtually identical to that of the total ALLW ($24,382 \text{ m}^3$ versus $24,413 \text{ m}^3$). Therefore, the concentrations of the nonactinides in the refined inventory for CH ALLW are reduced in proportion to their respective reductions in radioactivity.

Comparison of Table 2-7 with Table 2-3 shows that the activities of most radionuclides in the RH ALLW are slightly smaller. (One exception is Co-60, which is much smaller—only about 14% of its earlier value.) This result was expected because (a) the activity in the CH ALLW was addressed separately from the RH ALLW and (b) the volume (and activity) of the RH ALLW was assumed to be only 41%, rather than the 50% used previously, of the total for the RH waste.

The volume of the RH ALLW is only about 0.13% of the total ALLW (31 m^3 versus $24,413 \text{ m}^3$). Therefore, the concentrations of the nonactinides obtained using the refined assumptions are about $1/0.0013 = 770$ times larger than the values developed using the simplified method.

The refined assumptions result in the activity and the activity concentration of the nonactinides generally decreasing by factors of 20 to 30 (depending on the radionuclide) for the CH ALLW. This decrease would be expected to reduce the receptor dose substantially for the intruder scenarios.

For the RH ALLW, the refined assumptions result in little change in the nonactinide activity inventory but an increase by a factor of about 770 in the activity concentration. This increase would be expected to increase the receptor dose substantially for intruder scenarios in which penetration of the RH waste occurs.

2.4.4 Inventory with Additional Refinements for Nonactinides

During the third phase of the study, a nonactinide "inventory with additional refinements" was developed and a few computer runs were made using this inventory. The inventory of actinides was not changed from that in Table 2-3.

Table 2-6. Refined inventory of nonactinides for CH ALLW.

Radionuclide	Radioactivity (Ci)
Ba-137m	54.3
Co-60	29.6
Cs-137	54.3
Sr-90	30.9
Y-90	30.9
Total	200

Table 2-7. Refined inventory of nonactinides for RH ALLW.

Radionuclide	Radioactivity (Ci)
Ba-137m	882
Co-60	6.8
Cs-137	882
Sr-90	798
Y-90	798
Total	3,367

Note: Totals may not match exactly because of rounding.

The details of developing the inventory with additional refinements are given in Section A-10 of Appendix A. An overview is given below. As with the refined inventory, the CH waste and RH waste were addressed separately.

Two investigations provided the new information for the inventory with additional refinements:

- One investigation was directed toward obtaining a more accurate split of the nonactinides between the TRU waste and the ALLW, and between the CH waste and the RH waste. The two previous inventories were based on very broad assumptions

concerning these questions. In the new investigation, the RWMIS entries for each waste shipment containing nonactinides totaling >1 Ci were examined to determine whether the containers in the shipment would be expected to be TRU waste or ALLW, and CH waste or RH waste. To be conservative, any shipments with a TRU activity <200 nCi/g were included in the ALLW. The division into the CH waste and RH waste was accomplished using separate RWMIS printouts for the CH waste on the TSA pads and the RH waste in the ILTSF. All individual nonactinide entries >0.1 Ci were then summed over all such shipments. Based on these new determinations, revisions were made to the estimate of the ALLW nonactinide inventory.

- In the other investigation, additional information on the nonactinide inventory was obtained from sources other than RWMIS. Based on various letters and reports, the following nonactinide activity (as of the time of disposal) that is not in RWMIS was identified:
 - As much as 149 Ci of H-3 in waste from Rocky Flats Plant (RFP)
 - An additional 64 Ci of Co-60 and 10 Ci of H-3 in radiation sources from RFP.

In addition, an estimate was made of the ALLW activity of three nonactinides that are often not reported or are underreported because of measurement difficulties. The three nonactinides—C-14, Tc-99, and I-129—are widespread in radioactive waste from non-weapons-related operations. Though very small in relative activity, they are often of interest in PAs because of their very long half-lives and relatively high mobility in the environment, once released from confinement. Using scaling factors derived for the buried waste at the SDA (Lockheed Idaho 1995), the following conservative estimates were made of the activities of these actinides:

- 1.0 Ci of C-14
- 0.05 Ci of Tc-99
- 1.8E-05 Ci of I-129.

The results of the above three efforts were subjected to screening to eliminate nonactinides of very small activity that will not affect the calculated doses, then were decayed to 1993. The product is the nonactinide inventory with additional refinements. The CH inventory, RH inventory, and combined inventory are presented in Tables 2-8, 2-9, and 2-10, respectively.

For comparison purposes, the corresponding entries from the simplified inventory and the refined inventory are also given in Tables 2-8 through 2-10. Table 2-10 shows that, with each successive inventory, the trend in total nonactinide activity has been downward.

2.4.5 Refined Modeling of Untreated Waste

In the second phase of the study, the performance of the untreated waste was evaluated as a benchmark for comparison. This necessitated developing a model of the untreated waste, in terms of the radionuclide inventory and release mechanisms.

Table 2-8. Nonactinide inventory with additional refinements for CH ALLW.

Radionuclide	Activity in inventory with additional refinements (Ci)	Activity in refined inventory ^a (Ci)
Ba-137m	67.3	54.3
C-14	1.0	—
Co-60	8.1	29.6
Cs-137	67.3	54.3
H-3	55.8	—
I-129	1.8E-5	—
Sr-90	54.1	30.9
Tc-99	0.05	—
Y-90	54.1	30.9
Total	307.7	200

a. Taken from Table 2-6.

Table 2-9. Nonactinide inventory with additional refinements for RH ALLW.

Radionuclide	Activity in inventory with additional refinements (Ci)	Activity in refined inventory ^a (Ci)
Ba-137m	140.5	882
C-14	1.0	—
Co-60	5.82	6.8
Cs-134	5.5	—
Cs-137	140.5	882
I-129	1.8E-5	—
Sr-90	100.2	798
Tc-99	0.05	—
Y-90	100.2	798
Total	493.8	3366.8

a. Taken from Table 2-7.

Table 2-10. Nonactinide inventory with additional refinements for CH plus RH ALLW.

Radionuclide	Activity in inventory with additional refinements (Ci)	Activity in refined inventory ^a (Ci)	Activity in simplified inventory ^b (Ci)
Ba-137m	207.8	936.3	1125
C-14	1.0	—	—
Co-60	13.92	36.4	47
Cs-134	5.5	—	—
Cs-137	207.8	936.3	1125
H-3	55.8	—	—
I-129	1.8E-5	—	—
Sr-90	154.3	828.9	1013
Tc-99	0.05	—	—
Y-90	154.3	828.9	1013
Total	800.5	3566.8	4323

a. Taken from Tables 2-6 and 2-7.

b. Taken from Table 2-3.

NOTE: As discussed in Section 2.3, RCRA regulations are expected to require thermal treatment of most of the ALLW, followed by immobilization of the ash and other residuals. As a simplification, untreated waste was modeled here, rather than unimmobilized waste that had been through thermal treatment.

The model is referred to here as "refined" because it is slightly more sophisticated than the simplified one described in Section 2.4.2. However, it is still a very rudimentary model. It places all the untreated ALLW into three categories, with homogeneous properties within a category.

The details of the refined model for the untreated ALLW are given in Appendix B. An overview is provided below.

The refined radionuclide inventory used for modeling the untreated ALLW coincides with that presented in Table 2-3 for the actinides and Tables 2-6 and 2-7 (the refined inventory) for the nonactinides. Thus, the present discussion does not involve a new radionuclide inventory but a regrouping of the same inventory.

After treatment, the INEL ALLW is expected to be fairly homogeneous, depending on the operational practices at the treatment facility. By contrast, the untreated ALLW is highly heterogeneous in its physical, chemical, and radiological characteristics.

No waste description of manageable complexity for this study can describe the ALLW precisely. Therefore, an approximate model representing the radiological inventory and release characteristics of the waste was developed. Grouping the untreated waste into categories with similar characteristics was necessary, together with assuming homogeneity of the waste within a category.

It was decided to group the ALLW packages first by *INEL identification (ID) number*. For mixed ALLW, there are 18 INEL ID numbers—groups of *content codes*^d—that are based primarily on physical similarities. Examples of INEL ID numbers are those for cemented sludges (six content codes), resins (three), metals (five), and combustibles (seven). For nonmixed ALLW, there are eight INEL ID numbers.

The INEL ID numbers were then grouped into a smaller number of waste *categories*. The characteristics of the waste within each category were then evaluated and compared against the characteristics of the other categories.

Waste records in the form of detailed descriptions of the waste within a content code (Apel et al. 1993, supplemented by Apel et al. 1994 as necessary) were used to arrive at appropriate categories for modeling the waste. Those descriptions provide a large amount of information on the waste, although they are incomplete for many content codes, particularly concerning the radionuclide quantities and distributions.

As described in Appendix B, the data from Apel were used only for scaling the actinide content among content codes, ID numbers, and categories. The actinide inventory used was that developed in Appendix A and discussed in Section 2.4.2.2. The nonactinide activity used was that developed in Section A-9 of Appendix A and discussed in Section 2.4.3, as adapted for the untreated waste.

Table 2-11 summarizes the refined model of the untreated waste. Five ID numbers and part of another one relate to waste in the form of sludges, resins, and salts. They are combined into category 1: sludges, etc. For more than 99% of the actinide activity in this category, the release mechanism is assumed to be diffusion. The release parameters were assumed to match those for content code 2, second-stage sludge, the dominant content code in terms of actinide inventory for this category.

All but one of the 21 remaining ID numbers were grouped into category 2, the nonsludges. Most of the actinides in category 2 are located on the surfaces of paper, rags, and metals. For most of the radioactivity in this category, the release mechanism is surface washoff. That mechanism was assumed for all ALLW in this category.

d. Each container of waste stored at the INEL is categorized under a specific content code, which provides a general description of the containerized waste. There are approximately 90 content codes.

Table 2-11. Refined model of the untreated ALLW.

Category	Volume (m ³)	Actinide inventory (Ci)		Nonactinide inventory (Ci)		Release mechanism	Content code for other properties
1—Sludges, etc.	3,904	Am-241	214	Ba-137m	8.70	Diffusion	2 (second-stage sludge)
		Cm-244	1.28	Co-60	4.74		
		Pu-238	145	Cs-137	8.70		
		Pu-239	95.7	Sr-90	4.95		
		Pu-240	23.8	Y-90	4.95		
		Pu-241	384				
		U-233	2.08				
		U-238	0.0003				
2—Nonsludges	20,455	Am-241	1,124	Ba-137m	45.6	Surface washoff	330 (dry paper and rags)
		Cm-244	6.71	Co-60	24.9		
		Pu-238	760	Cs-137	45.6		
		Pu-239	502	Sr-90	25.9		
		Pu-240	123	Y-90	25.9		
		Pu-241	2,014				
		U-233	10.9				
		U-238	0.0014				
3—Remote-handled	31 ^a	Am-241	1.647	Ba-137m	882	Surface washoff	None appropriate; assume particulate on filters or rags
		Cm-244	0.010	Co-60	6.8		
		Pu-238	1.15	Cs-137	882		
		Pu-239	0.735	Sr-90	798		
		Pu-240	0.181	Y-90	798		
		Pu-241	2,950				
		U-233	0.016				
		U-238	0				
Total	24,390 ^a	Am-241	1,340	Ba-137m	936	NA	NA
		Cm-244	8	Co-60	36.4		
		Pu-238	906	Cs-137	936		
		Pu-239	598	Sr-90	829		
		Pu-240	147	Y-90	829		
		Pu-241	2,401				
		U-233	13				
		U-238	0.0017				

a. For consistency with other parametric studies in this document, the volume of the RH waste was assumed to be 31 m³, as given in RWMIS and discussed in Appendix A, rather than the 54 m³ given in Raivo et al. 1993. The total volume over all categories is, therefore, 23 m³ less than the 24,413 m³ in Raivo et al. 1993. This volume difference of less than 0.1% is not significant for developing the radiological inventory and was neglected.

The last ID number contains all of the RH ALLW. The waste with this ID number was assigned to category 3, the RH waste. The available information on physical and chemical form is not detailed; a surface washoff release mechanism was assumed because most of the waste is combustibles and (apparently) nonactivated metals.

2.5 Radionuclide Screening of Nonactinides from Detailed Groundwater Pathway Evaluation

2.5.1 Objective and Scope

The calculations of groundwater transport of radionuclides using the models applied in this study are complex. In addition, separate model runs are required for each radionuclide whose contribution to the total dose is evaluated. Therefore, simple screening calculations were performed. The objective was to eliminate from detailed groundwater transport calculations the radionuclides that could make only negligible contributions to the dose.

This section describes how the screening was applied to the nonactinides in each of the three radionuclide inventories: simplified, refined, and with additional refinements. All of the actinides, the activities of which are identical in the three inventories, were retained for the detailed evaluations of groundwater transport.

This screening applies only to the groundwater pathway. Radionuclides identified as being insignificant dose contributors via that pathway may still make important contributions to dose in the intrusion scenarios. Therefore, the complete inventories in Table 2-3, in Tables 2-6 and 2-7, and in Tables 2-8 through 2-10 were retained for the evaluation of the intruder scenarios.

2.5.2 Approach and Generic Assumptions

The screening model of Seitz and Kocher (1993) was used to establish trigger levels for radionuclide concentrations in the waste. The trigger levels indicate what constitutes a potentially important concentration of a given radionuclide in the FWF, based on potential groundwater concentrations that could exceed drinking water standards established by the Nuclear Regulatory Commission (NRC) in 10 CFR Part 20 (NRC 1989). If the concentration of a given radionuclide in the FWF is below the trigger level, then that radionuclide will not make an important contribution to dose. Further calculations or data-collection efforts are not required for that radionuclide.

The starting point for the analysis was the following screening equation proposed by Seitz and Kocher (1993):

$$T_i = \frac{D_{std_i} \theta R_{d_i}}{D_{ing_i} C_r e^{-\lambda_i t}} \quad (2-1)$$

where

- T_i = trigger level for radionuclide i (Ci/m^3)
- D_{std_i} = dose limit for radionuclide i (mrem/yr)
- D_{ing_i} = ingestion dose conversion factor for radionuclide i (mrem/Ci)
- C_r = consumption rate (m^3/yr)
- t = time at which consumption takes place (yr)
- θ = porosity of the aquifer (dimensionless)
- R_{d_i} = retardation coefficient for radionuclide i in the aquifer (dimensionless)
- λ_i = decay constant for radionuclide i ($1/\text{yr}$) = $(\ln 2)/t_{1/2i}$ with $t_{1/2i}$ = half-life of radionuclide i (yr).

Equation (2-1) assumes that the entire volume of the waste inventory is deposited in an equal volume of aquifer at time t . The factor $D_{std_i}/(D_{ing_i} C_r)$ is equivalent to a concentration limit of radionuclide i in the groundwater. This factor, or concentration limit, will be denoted by the variable C_i and has the units of Ci/m^3 .

The following generic assumptions were used for the screening on all three inventories:

1. No credit was taken for any radionuclide immobilization provided by the FWF. The approach assumed that the entire radionuclide inventory is initially dissolved in the pore water within the disposal unit.
2. The time, t , was conservatively assumed to be the product of the vadose zone pore water travel time and the retardation coefficient for each radionuclide.
3. Conservatively, the value of R_{d_i} listed explicitly in Equation (2-1) was set to the minimum value of 1.0.
4. The concentration limit of radionuclide i in the groundwater, C_i , was assumed to be equal to the smallest value of the maximum permissible concentration (MPC) for public drinking water listed for radionuclide i in "Standards for Protection Against Radiation," Appendix B in 10 CFR Part 20 (NRC 1989).

5. The half-life of Y-90 (64 hr) was conservatively assigned the value for the half-life of Sr-90 (about 29 yr) in order to maintain secular equilibrium with the progenitor radionuclide. The same approach was used with Ba-137m and Cs-137.
6. The concentration of radionuclides in the FWF was conservatively selected to be equal to the concentration in IEB4 (the more favorable ceramic FWF), since this FWF has the highest waste loading.

2.5.3 Screening Calculations for the Simplified Inventory

During the first phase of the study, the screening calculations were performed on the simplified nonactinide inventory in Table 2-3. The calculations were applied to groundwater transport at Sites A, B, and C.

A groundwater travel time of 400 yr was used. This value was the smallest value for any of the three sites, as shown later in Section 3. An aquifer porosity of 0.1 was used. The retardation coefficient R_d for each radionuclide was calculated conservatively by using (a) one-tenth of the smaller K_d values listed later in Table 3-2 and (b) a bulk density of 1.5 g/cm^3 and porosity of 0.06 for the vadose zone, where nearly all of the travel time occurs.

By means of the screening method just described, the five nonactinides in the simplified inventory, Co-60, Y-90, Sr-90, Ba-137m, and Cs-137, were determined to be insignificant dose contributors via the groundwater pathway. They were deleted from detailed analysis for that pathway. The remaining radionuclides (the actinides) were analyzed in detail using the groundwater transport model, because they are progenitors of long-half-life decay chains.

Table 2-12 shows the bases for those conclusions. The second column lists the concentration of each nonactinide radionuclide in the FWF, as taken from Table 2-5. The third column lists the radionuclide half-lives. The fourth column lists the decay constant (λ_i) for each radionuclide.

The fifth column in the table lists the MPC in drinking water, as stipulated in 10 CFR Part 20 (NRC 1989).

Table 2-12. Screening of nonactinides from the simplified inventory before detailed evaluation of groundwater transport—Sites A, B, and C.

Radionuclide	Concentration in FWF (Ci/m ³)	Half-life (yr)	Lambda (1/yr)	MPC drinking water (Ci/m ³)	Trigger level (Ci/m ³)	Concentration in FWF/trigger level
Co-60	8.27E-03	5.3	0.131	3E-05	>1E+200	<1E-200
Sr-90	1.78E-01	29	0.0239	3E-07	6.7E+17	7.4E-19
Y-90	1.78E-01	29 ^a	0.0239	2E-05	4.5E+19	1.1E-20
Ba-137m	1.98E-01	30.2 ^a	0.0230	2E-05	>1E+200	<1E-200
Cs-137	1.98E-01	30.2	0.0230	2E-05	7.0E+197	2.8E-199

a. See text for explanation of half-life used.

The sixth column lists the calculated trigger level for each radionuclide, with the variable C replacing the ratio $D_{std}/(D_{ing} C_r)$, using the values of the variables listed earlier. Entries in the sixth column designated by $>1E+200$ indicate that the denominator approaches zero, based on the large values for the time (t) for the radionuclide to reach the saturated zone relative to the half-life. Since the denominator approaches zero, the trigger level approaches very large values, and the radionuclide is not of concern. The final column lists the ratio of the radionuclide concentration in the FWF to that of the trigger level for each radionuclide. If a value is greater than 1.0, that radionuclide justifies detailed evaluation of groundwater transport. All values listed are many orders of magnitude less than 1.0, so further groundwater transport evaluation of the nonactinides in this inventory was not necessary.

The additional nonactinides Ce-144, Cs-134, Kr-85, Ni-63, Pm-147, and Pr-144 that were excluded initially from the simplified inventory, in Table A-2, based on experience, were subjected later to the above screening analysis. The results showed that these radionuclides, at their assumed inventories in Table A-2, do not contribute substantially to the dose for the groundwater pathway.

These analyses show that the nonactinides in the simplified inventory would not contribute substantially to the dose via groundwater transport in this study. These radionuclides were retained, however, for the analysis of the intrusion scenarios.

As will be shown later in Section 10, a limited number of runs were made with groundwater travel times considerably shorter than the original values. For Sites A, B, and C, the shortest time was 6.3 yr for some Site A runs. Overall, the shortest time was 1.5 yr for one Site D run. The question arises as to whether the nonactinide screening conclusions drawn from Table 2-12, which was based on a 400-yr travel time, are still valid for the shorter travel times. Part of the answer is presented in Table 2-13. That table also includes the very small activities of C-14, Tc-99, I-129, and H-3 given in RWMIS, as taken from Table A-2. For the minimum groundwater travel time of 6.3 yr for Sites A, B, and C, the nonactinides Sr-90, Y-90, and Cs-137 now pass the screening.

Table 2-13. Screening of nonactinides from the simplified inventory before detailed evaluation of groundwater transport—Sites A, B, and C with minimum groundwater travel time of 6.3 yr.

Radionuclide	Concentration in FWF (Ci/m ³)	Half-life (yr)	Lambda (1/yr)	MPC drinking water (Ci/m ³)	Trigger level (Ci/m ³)	Concentration in FWF/trigger level
Co-60	8.27E-03	5.3	0.131	3E-05	2.7E+84	3.1E-87
Sr-90	1.78E-01	29.0	0.0239	3E-07	7.3E-08	2.4E+06
Y-90	1.78E-01	29.0	0.0239	2E-05	4.9E-06	3.6E+04
Ba-137m	1.98E-01	30.2	0.0230	2E-05	6.5E+03	3.1E-05
Cs-137	1.98E-01	30.2	0.0230	2E-05	3.2E-03	62
C-14	<1.76E-08	5,720	1.21E-04	8E-04	8.0E-05	<2.2E-04
Tc-99	1.58E-07	2.13E+05	3.25E-06	2E-04	2.0E-05	7.9E-03
I-129	0	1.7E+07	4.08E-08	6E-08	6.0E-09	0
H-3	7.04E-05	12.3	5.6E-02	3E-03	4.3E-04	0.16

Detailed groundwater runs were made for these three nonactinides to determine whether the maximum dose would be affected by the nonactinides in cases involving the shortest travel times. No maximum doses were affected.

Screening of the simplified inventory for application at Sites D and E is discussed at the end of Section 2.5.4 below. That section pertains to screening of the inventory with additional refinements, which was developed in the third phase of the study.

2.5.4 Screening Calculations for the Inventory with Additional Refinements

In the third phase of the study, the same screening method was applied to the inventory with additional refinements. The combined CH and RH inventory in Table 2-10 was used.

Except as discussed here, the same screening assumptions were used as in Section 2.5.3.

The inventory with additional refinements was used for a limited number of runs at Sites A, C, D, and E. Therefore, the screening considered all of these sites with their respective groundwater travel times and sorptive behaviors. Of the sites, the most favorable in terms of minimal groundwater transport is Site C and the least favorable is Site D. The other sites fall between the results for these two sites.

As will be discussed later in Section 3, for Site D, the groundwater travel time is 4.52 yr. For Site C, it is 2246 yr. These travel times are multiplied by the retardation coefficient of each radionuclide, which is calculated from the K_d s.

The results of the screening are shown in Table 2-14 for the most favorable site and in Table 2-15 for the least favorable site.

The Cs-137, Ba-137m, Co-60, Sr-90, and Y-90 were screened out at both sites. The Cs-134 also was screened out at both sites. Because H-3, C-14, Tc-99, and I-129 could not be screened out, they were retained for detailed groundwater transport evaluations at all sites in the runs using this inventory. These radionuclides exceed the screening threshold at the least favorable site by factors ranging from 5.5 to 320. Retention in the screening does not necessarily imply, however, that these radionuclides will be substantial contributors to dose in the detailed calculations.

Again, all of the radionuclides in the inventory were retained for the evaluations of the intrusion scenarios.

With completion of the screening in Table 2-15 for the inventory with additional refinements, the question of using the screened simplified inventory for runs at Sites D and E can now be addressed. The question is whether radionuclides in the simplified inventory that were screened out of detailed groundwater evaluations based on the characteristics of Sites A, B, and C should be included in the simplified inventory when applied to Sites D and E. The question was addressed by means of Table 2-15, combined with Table 2-5 and the unscreened ALLW inventory column in Table A-2, which is the precursor to Table 2-3.

Table 2-14. Screening of nonactinides from the inventory with additional refinements before detailed evaluation of groundwater transport—Site C (most favorable).

Radionuclide	Concentration in FWF (Ci/m ³)	Half-life (yr)	Lambda (1/yr)	MPC drinking water (Ci/m ³)	Trigger level (Ci/m ³)	Concentration in FWF/trigger level
C-14	1.76E-04	5720	1.21E-04	8E-04	8.4E-06	21
I-129	3.16E-09	1.7E07	4.08E-08	6E-08	4.8E-10	6.6
Tc-99	8.79E-06	2.12E05	3.25E-06	2E-04	1.6E-06	5.4
Ba-137m	3.65E-02	30.2	2.3E-02	3E-08	>E+200	<E-200
Co-60	2.45E-03	5.3	0.13	3E-05	>E+200	<E-200
Cs-134	9.74E-04	2.1	0.34	9E-06	>E+200	<E-200
Cs-137	3.65E-02	30.2	2.3E-02	2E-05	>E+200	<E-200
Sr-90	2.71E-02	29.0	2.4E-02	3E-07	>E+200	<E-200
Y-90	2.71E-02	29.0	2.4E-02	2E-05	>E+200	<E-200
H-3	9.81E-03	12.3	5.6E-02	3E-03	1.8E+50	5.4E-53

Table 2-15. Screening of nonactinides from the inventory with additional refinements before detailed evaluation of groundwater transport—Site D (least favorable).

Radionuclide	Concentration in FWF (Ci/m ³)	Half-life (yr)	Lambda (1/yr)	MPC drinking water (Ci/m ³)	Trigger level (Ci/m ³)	Concentration in FWF/trigger level
C-14	1.76E-04	5720	1.21E-04	8E-04	6.5E-06	27
I-129	3.16E-09	1.7E+07	4.08E-08	6E-08	4.8E-10	6.6
Tc-99	8.79E-06	2.13E+05	3.25E-06	2E-04	1.6E-06	5.5
Ba-137m	3.65E-02	30.2	2.3E-02	3E-08	2.8E+33	1.3E-35
Co-60	2.45E-03	5.3	0.13	3E-05	>E+200	<E-200
Cs-134	9.74E-04	2.1	0.34	9E-06	>E+200	<E-200
Cs-137	3.65E-02	30.2	2.3E-02	2E-05	9.1E+64	4.0E-67
Sr-90	2.71E-02	29.0	2.4E-02	3E-07	7.7E-02	0.35
Y-90	2.71E-02	29.0	2.4E-02	2E-05	5.1	5.3E-03
H-3	9.8E-03	12.3	5.6E-02	3E-03	3.1E-05	320

The results are given in Table 2-16. This table is based on the simplified inventory of nonactinides in Table A-2, including the very small activities of C-14, Tc-99, I-129, and H-3 given in RWMIS. Table 2-16 indicates that Sr-90 and H-3 survive the screening at Site D. As was the case with Sites A, B, and C, detailed groundwater runs with the simplified inventory were made for Site D using Sr-90 and H-3 and the shortest travel times. Again, the maximum doses were not affected by the nonactinides.

2.5.5 Screening Calculations for the Refined Inventory

No separate screening was done for the refined inventory. Instead, the conclusions drawn for the simplified inventory and for the inventory with additional refinements were applied, based on the following rationale.

Table 2-16. Screening of nonactinides from the simplified inventory before detailed evaluation of groundwater transport—Site D (least favorable).

Radionuclide	Concentration in FWF (Ci/m ³)	Half-life (yr)	Lambda (1/yr)	MPC drinking water (Ci/m ³)	Trigger level (Ci/m ³)	Concentration in FWF/trigger level
C-14	1.76E-08	5720	1.21E-04	8E-04	6.5E-06	<2.7E-03
I-129	0	1.7E+07	4.08E-08	6E-08	4.8E-10	0
Tc-99	1.58E-07	2.13E+05	3.25E-06	2E-04	1.6E-06	0.099
Ba-137m	1.98E-01	30.2	2.3E-02	3E-08	2.8E+33	7.1E-35
Co-60	8.27E-03	5.3	0.13	3E-05	>E+200	<E-200
Cs-134	9.68E-03	2.1	0.34	9E-06	>E+200	<E-200
Cs-137	1.98E-01	30.2	2.3E-02	2E-05	9.1E+64	2.2E-66
Sr-90	1.78E-01	29.0	2.4E-02	3E-07	7.7E-02	2.3
Y-90	1.78E-01	29.0	2.4E-02	2E-05	5.1	0.035
H-3	7.04E-05	12.3	5.6E-02	3E-03	3.1E-05	2.3

Tables 2-6 and 2-7 present the refined inventory. The groundwater pathway evaluation was done for the combined CH plus RH waste; thus, for each nonactinide, the sum of the entries on the two tables is of interest. For each of the five nonactinides listed in those two tables, the total activity is less than that for the simplified inventory in Table 2-3. Because those five radionuclides were screened out from detailed groundwater transport evaluation in Table 2-12 and Section 2.5.3, the smaller activities in the refined inventory were also screened out.

The activities of Cs-137, Sr-90, and Y-90 in the refined inventory are smaller than in the simplified inventory. Detailed groundwater runs with the simplified inventory showed that these three radionuclides did not affect the maximum doses for the cases with the shortest travel times. Therefore, they would not be expected to affect the maximum doses for the refined inventory.

References for Section 2

- Apel, M. L., M. T. Muneta-Bradford, A. L. Robert, J. C. Tozer, M. D. Argyle, and G. K. Becker, 1993, *Waste Description Information Package for Alpha Low-level Mixed Waste and Alpha Low-level Waste (draft)*, Attachment 2 to *Solid Alpha-Contaminated Low-Level Radioactive Waste Transport/Treatment Services Request for Proposal*, SP-SOW-01, July.
- Apel, M. L., G. K. Becker, Z. K. Ragan, J. Frasure, B. D. Raivo, L. G. Gale, and D. P. Pace, 1994, *Radiological, Physical, and Chemical Characterization of Low-level Alpha-Contaminated Wastes Stored at the Idaho National Engineering Laboratory for Use in Conducting Feasibility Studies for Treatment Services*, EG&G Idaho, EGG-RWMC-11189, March.
- Atwood, C. L., and M. J. Schlafman, 1993, *Empirical Distributions of Radionuclides from RWMIS Data*, EG&G Idaho, Inc., EGG-RAAM-10741, April.
- EG&G Idaho, 1985, *A History of the Radioactive Waste Management Complex at the Idaho National Engineering Laboratory*, WM-F1-81-003, Rev. 3, July.
- Lockheed Idaho, 1995, *A Comprehensive Inventory of Radiological and Nonradiological Contaminants in Waste Buried in the Subsurface Disposal Area of the INEL RWMC During the Years 1984-2003*, Lockheed Idaho Technologies Company, INEL-95/0135, May.
- NRC, 1989, "Standards for Protection Against Radiation," 10 CFR 20, Appendix B, U.S. Nuclear Regulatory Commission.
- Raivo, B. D., T. L. Eddy, and G. L. Anderson, 1993, *Statement of Work: Commercial Transport and Treatment of Alpha-Contaminated Mixed Low-Level Radioactive Waste (draft)*, Attachment 1 to *Solid Alpha-Contaminated Low-Level Radioactive Waste Transport/Treatment Services Request for Proposal*, SP-SOW-01, June.
- Seitz, R. R., and D. C. Kocher, 1993, "On Establishing Waste Concentration Limits for Low-level Radioactive Waste Disposal," *Proceedings of the Symposium on Waste Management, Tucson, Arizona*, March.
- Smith, T. H., 1993, "Comparison of Two Estimates of Radionuclide Inventories in Alpha LLW and Alpha LLMW," EG&G Idaho, Inc., letter to G. L. Anderson, THS-13-93, August.
- Tozer, J. C., 1993, "Estimated Total Quantity of Nuclides in ALLW and ALLMW," EG&G Idaho, Inc., letter to M. L. Apel, JCT-16-93, July.

7. EVALUATION OF PERFORMANCE IN UNDISTURBED-WASTE SCENARIO—EXPOSURE ROUTES FOR GROUNDWATER TRANSPORT PATHWAY

7.1 Introduction

For the assessment of the groundwater transport pathway in the undisturbed-waste scenario, the exposure was assumed to occur by the use of groundwater from a well drilled 100 m from the downgradient edge of the disposal facility. (No disturbance of the waste zone occurs in this scenario.) The assumed well screen thickness is 15 m. The well is used for drinking water, irrigating crops, and watering livestock that may be consumed by humans.

The transport pathways and exposure routes for the undisturbed-waste scenario are depicted in the lower portion of Figure 4-2. Several pathways lead to the ingestion exposure route. (Section 4.2.1 discussed why other transport pathways and exposure routes were considered for this scenario but were not evaluated. One exception is radon transport, which is discussed in Section 8.) The details of the evaluation of the ingestion exposure route are given in Section 7.2.

An additional groundwater receptor location was evaluated in connection with Site A. A well was assumed to be drilled 215 km downgradient for one of the simulations (Run 1LDW). At that location, potentially contaminated groundwater may discharge to springs or other surface-water bodies. The same transport pathways and exposure routes were evaluated as for the 100-m well.

7.2 Ingestion

7.2.1 Conceptual Model

The conceptual model of the ingestion exposure route for the groundwater transport pathway assumed that a receptor resides near the disposal facility and uses a well located 100 m downgradient from the edge of the disposal unit as the major source of water. The receptor drinks contaminated groundwater, eats leafy vegetables and produce irrigated with contaminated groundwater, and consumes milk, meat, and eggs from livestock and poultry that consume contaminated groundwater and pasture grass irrigated with contaminated groundwater.

Secondary and indirect pathways within the groundwater pathway, such as inhalation of contaminated irrigation water and inhalation of contaminated dust remaining after evaporation of the irrigation water, were omitted from the conceptual model. These pathways would contribute relatively minor doses compared with those from more direct pathways, such as direct ingestion of contaminated water.

7.2.2 Numerical Model

The GENII dose assessment software package (Napier et al. 1988) was used to implement the conceptual model and to perform dose calculations based on the methods recommended by the International Committee on Radiation Protection (ICRP), as described in publications

ICRP 26 and 30. The long-term environmental dosimetry portion of the package consists of the DITTY code, which calculates population exposures from waste disposal sites over long time periods. The timeframe for the calculations in DITTY is any 10,000-yr period. This period is divided into 143 periods of 70 yr each. The DITTY code calculates the lifetime cumulative EDE and dose to critical organs for each time period and the annual EDEs for the period of maximum dose. The activity present during any period is the sum of the material released during that period plus the residual material in the environment from releases in previous periods.

The use of the DITTY code is adequate for calculating the maximum annual EDE within a 10,000-yr time-of-compliance period and comparing the predictions with the 25-mrem/yr performance objective. However, because the decay and in-growth of long-lived radionuclides may exceed the 10,000-yr time period, the maximum dose may be realized hundreds of thousands to millions of years after facility closure. For the analyses presented in this report, the maximum dose has been calculated irrespective of a time-of-compliance period. These maximum doses were calculated through the following sequence of calculations:

- Based on a conservative screening model (discussed in Section 2.5), several of the shorter-lived radionuclides were eliminated from further detailed analyses.
- The DUST code predicted the source terms for long-lived radionuclides (discussed in Section 6.2).
- The WELLC code predicted concentrations of the progenitor radionuclides at the receptor well (discussed in Section 6.4).
- The DECAYDOS program calculated the decay and in-growth of each radionuclide using the Bateman decay equations. Since the concentrations of the parent and progeny were calculated at the receptor well, this assumes identical sorption coefficients (and hence transport rates) for parent and progeny. Although radionuclides with lower sorption coefficients would arrive at the receptor well earlier, the annual EDE would be less than for the conservative assumed case of parent and progeny arriving at the receptor well simultaneously. Using as input the time-dependent concentration of each progenitor and progeny radionuclide and the groundwater usage environmental parameter values discussed in Section 7.2.3, the DITTY code calculated an annual EDE (in mrem/yr) for each radionuclide. The environmental parameter values assumed that the well water was used as the source of drinking water, as well as the source of irrigation water. The values used are summarized in Section 7.2.3, while the effective dose conversion factors for each radionuclide are summarized in Section 7.2.4.
- The doses were summed across all radionuclides that are part of a single decay chain. Thus, a dose-versus-time curve was derived for each chain.

7.2.3 Assumptions and Parameter Values

The required environmental parameter values encompass food-chain parameters and consumption rates. The parameter values used to evaluate the undisturbed-waste scenario with the DITTY code of the GENII package are provided in Table 7-1. Every effort was made to

Table 7-1. Parameter values for the DITTY code of the GENII package used for undisturbed-waste scenario.

Parameter	Value
Drinking water consumption rate	730.5 L/yr ^a
Consumption rate of leafy vegetables	11 kg/yr ^a $\times$ 0.25 ^b = 2.8 kg/yr
Consumption rate of other vegetables	166 kg/yr ^a $\times$ 0.25 ^b = 42 kg/yr
Consumption rate of eggs	10 kg/yr ^a
Consumption rate of milk	103 kg/yr ^a
Consumption rate of beef	59 kg/yr ^a
Consumption rate of pork	30 kg/yr ^a
Consumption rate of poultry	9 kg/yr ^a
Growing period for leafy vegetables	45 days ^a
Growing period for other vegetables	90 days ^a
Grain growing time for animal consumption	45 days ^a
Duration irrigation is used on crops	6 months/yr ^a
Irrigation rate for each crop	63.33 L/m ² /yr ^a

a. Source: Kennedy and Strenge 1992.

b. Based on the results of a U.S. Department of Agriculture survey, only 25% of the vegetables in the exposed individual's diet is derived from a residential garden (Aaberg and Kennedy 1990, Kennedy and Strenge 1992).

ensure that the parameter values supplied as input have referenceable sources to the extent of the available literature values. The majority of the parameter values were based on those cited by Kennedy and Strenge (1992). The values provided in that report are based on extensive U.S. Department of Agriculture consumption studies. Values for the other parameters are based on the recommended values supplied in the GENII package.

7.2.4 Effective Dose Conversion Factors for Groundwater Pathway

Table 7-2 presents the effective dose conversion factors for the groundwater pathway for the various radionuclides used in DECAYDOS. The effective dose conversion factors were obtained by running the DITTY routine of GENII with a unit concentration (1 Ci/m³) of each radionuclide in well water and with the parameter values in Table 7-1. These effective dose conversion factors differ from standard ingestion dose conversion factors in that they sum the doses from multiple environmental pathways. Included are the ingestion of 2 L/day of water, the consumption of leafy and other vegetables irrigated with contaminated water, and the consumption of vegetables, beef, poultry, pork, eggs, and milk obtained from animals that consumed contaminated water.

Table 7-2. Effective dose conversion factors for the various radionuclides used in DECAYDOS.

Radionuclide	Effective dose conversion factor (rem/yr)/(Ci/m ³)	Radionuclide	Effective dose conversion factor (rem/yr)/(Ci/m ³)
Am-241	1.7×10^6	Ac-227	6.9×10^6
Pu-240	3.4×10^4	Ac-225	7.0×10^4
Pu-239	3.4×10^4	Ra-228	5.1×10^5
Pu-238	3.1×10^4	Ra-226	7.7×10^5
Np-237	1.3×10^7	Ra-225	1.4×10^5
U-238	1.9×10^4	Ra-224	7.9×10^4
U-236	2.0×10^4	Ra-223	1.7×10^5
U-235	2.7×10^4	Rn-222	1.6×10^1
U-234	2.1×10^4	Bi-212	9.2×10^2
U-233	2.1×10^4	Bi-210	5.6×10^3
Th-234	8.7×10^3	Pb-212	3.8×10^4
Th-232	1.3×10^6	Pb-210	3.8×10^6
Th-231	8.3×10^2	Po-210	1.4×10^6
Th-230	2.8×10^5	Cs-137	8.5×10^4
Th-229	1.8×10^6	I-129	3.5×10^5
Th-228	2.5×10^5	Tc-99	4.5×10^3
Th-227	2.5×10^4	Y-90	2.0×10^3
Pa-234	1.4×10^3	Sr-90	2.4×10^5
Pa-233	2.4×10^3	C-14	7.5×10^3
Pa-231	6.2×10^6	H-3	7.0×10^1
Ac-228	1.3×10^3	—	—

These factors yield doses in rem/yr for each radionuclide, based on the use of groundwater with a concentration of 1 Ci/m³ for each radionuclide. To provide actual doses, the factors need to be adjusted for the actual concentration of each radionuclide in the groundwater.

The standard dose conversion factors upon which these effective dose conversion factors are based are from the GENII library, which uses the most conservative dose conversion factors contained in DOE 1988a and DOE 1988b.

References for Section 7

- Aaberg, R. L., and W. E. Kennedy, 1990, *Definition of Intrusion Scenarios and Example Concentration Ranges for the Disposal of Near-Surface Waste at the Hanford Site*, PNL-6312, Pacific Northwest Laboratories.
- DOE, 1988a, *External Dose-Rate Conversion Factors for Calculation of Dose to the Public*, DOE/EH-0070, U.S. Department of Energy, March.
- DOE, 1988b, *Internal Dose-Rate Conversion Factors for Calculation of Dose to the Public*, DOE/EH-0071, U.S. Department of Energy, July.
- Kennedy, W. E., and D. L. Streng, 1992, *Residual Radioactive Contamination from Decommissioning: Technical Basis for Translating Contamination Levels to Annual Total Effective Dose Equivalent*, NUREG/CR-5512, Pacific Northwest Laboratory.
- Napier, B. A., R. A. Peloquin, D. L. Streng, and J. V. Ramsdell, 1988, *GENII-The Hanford Environmental Radiation Dosimetry Software System: Conceptual Representation*, PNL-6584, Vol. 1, Pacific Northwest Laboratory.

8. EVALUATION OF PERFORMANCE IN UNDISTURBED-WASTE SCENARIO—ATMOSPHERIC TRANSPORT PATHWAY (RADON TRANSPORT AND RECEPTOR EXPOSURE)

8.1 Introduction

Section 4.2.1 discussed why the doses from other atmospheric pathways in the undisturbed-waste scenario were considered negligible compared with those from the transport of gaseous radionuclides. Radon from the decay of actinides is expected to be the only gaseous radionuclide meriting evaluation and is discussed in this section. Other gaseous radionuclides are present (e.g., Kr-85), but the activities are extremely small. In addition, those radionuclides either decay relatively quickly or will tend to partition into the gas phase during thermal treatment and not be incorporated into the FWF.

Three radionuclides of radon, Rn-219, Rn-220, and Rn-222, will be produced as decay products of actinides in the inventory. Rn-222, which grows in from the decay of Pu-238 and U-238 in the inventory, has a half-life of about 3.8 days, the longest half-life of the three radon radionuclides. Production rates of Rn-222 were calculated using the DECAYDOS routine, with the conservative assumption that no leaching of the parent radionuclides by groundwater occurs.

In the assumed absence of leaching, the short-lived radon produced by decay within a nonporous FWF (such as glass or ceramic) will not exist long enough as a gas for transport to the land surface. Only the radon produced from parent actinides that are in a porous FWF or are already released from the FWF has the potential to exist long enough to be transported to the surface. Therefore, only the transport of Rn-222 from a cement FWF was evaluated.

The remainder of Section 8 describes the evaluation of the transport of Rn-222 and the resulting dose from the inhalation of Rn-222 and its short-lived decay products (Po-218, Pb-214, Bi-214, and Po-214) that are generated post-inhalation. The approach was to develop a conceptual model of the transport pathway and the exposure route. An analytical solution was then used to implement the conceptual model.

8.2 Conceptual Model

Figure 8-1 presents a sketch of the assumed transport pathway for radon exposure in the undisturbed-waste scenario. The only exposure route is inhalation.

The radon is produced by the radioactive decay of Pu-238 and U-238 contained in a PCC FWF within a 22.7-ft-thick layer of waste. Conservatively, the more shallow of the two burial depths was assumed. That is, the waste is overlain by a 16-ft layer of soil (5 ft of belowgrade backfill plus 11 ft of abovegrade cover that is conservatively assumed to be all soil). The abovegrade cover erodes instantaneously at a time 500 yr postclosure. One dose calculation was performed just before 500 yr postclosure, when the aboveground cover is still present. A second dose was calculated for a time 167,000 yr postclosure, when the radon concentration in the PCC FWF was determined by means of DECAYDOS to be a maximum.

Figure 8-1. Simplified depiction of conceptual model for radon transport and receptor exposure. (Until 500 yr postclosure, an 11-ft abovegrade cover was also assumed to be present.)

The radon diffuses through the air-filled porosity within the PCC to the FWF surface and then diffuses through the air-filled soil porosity toward the land surface. At the land surface, the radon is diluted and transported by air currents to a receptor located at the downwind edge of the waste pit. The receptor becomes exposed by breathing the contaminated air.

8.3 Numerical Model

For simplicity, an analytical solution was used to calculate the Rn-222 emissions. The flux of Rn-222 from the FWF was estimated based on the equation (Case et al. 1990):

$$J_o = 10,000 D_e C_{Rn} \left(\frac{\lambda \epsilon}{D_e} \right)^{1/2} \quad (8-1)$$

where

- J_o = radon flux from the FWF ($\text{pCi}/\text{m}^2 \cdot \text{sec}$)
- 10,000 = conversion factor (cm^2/m^2)
- D_e = effective diffusion coefficient in the FWF (cm^2/sec)

C_{Rn} = concentration of Rn-222 in the void spaces of the FWF, defined below (pCi/cm³)

λ = decay constant for Rn-222 (2.097×10^{-6} /sec)

ϵ = void fraction of the FWF.

The concentration of Rn-222 in the void spaces is related to the concentration in the waste form, $C_{Rn}(FWF)$, by the equation $C_{Rn}(\text{void space}) = C_{Rn}(FWF)/\epsilon$. The DECAYDOS program calculated the radon concentration $C_{Rn}(FWF)$ using the Bateman decay equations.

The effective diffusion coefficient, D_e , was calculated based on the equation $D_e = \epsilon D$, where D is the diffusion coefficient for radon in the air space of the FWF. D was calculated based on the diffusion coefficient for radon in air and the tortuosity of the PCC FWF, which was assumed to be 200.

Using the definitions of variables outlined above, equation (8-1) can be reduced to the following:

$$J_o = 10,000 C_{Rn} (\lambda D)^{1/2} \quad (8-2)$$

where

C_{Rn} is now the concentration of Rn-222 in the FWF itself.

The radon flux through the soil cover was calculated based on the equation:

$$J = J_o f_s e^{(-\alpha_s x_s)} \quad (8-3)$$

where

J = radon flux from the ground surface above the disposal facility (pCi/m² • sec)

J_o = radon flux from the waste (pCi/m² • sec)

f_s = correction factor for the soil cover (dimensionless)

α_s = $(\lambda \epsilon / D_e)^{1/2}$ for soil(s) (cm)⁻¹

x_s = thickness of soil cover (cm).

Given a radon flux from the ground surface above the disposal facility (J), the concentration of radon for a receptor directly above the disposal facility was calculated using an Eulerian box model, which is recommended for sites where the contaminant emission rate is relatively constant over an extended period of time (EPA 1991). For this application, the receptor was located at

the downwind edge of the disposal facility in order to maximize the radon concentration in the "box" reaching the receptor. The general equation for the box model is:

$$C_x = \frac{JL}{U\sigma_x} \quad (8-4)$$

where

- C_x = concentration of contaminant at the receptor (pCi/m³)
- J = flux of contaminant from the ground surface (pCi/m² • sec)
- L = travel distance (m)
- U = wind speed in the mixing zone (m/sec)
- σ_x = mean vertical mixing height (m).

8.4 Assumptions and Parameter Values

As mentioned previously, it was assumed that the thickness of the waste layer is 22.7 ft. The cover above the waste is a total of 16 ft thick until 500 yr postclosure, after which time the cover thickness is only the 5 ft of backfill.

The inventory of Rn-222 was calculated by decaying the 1993 inventories of Pu-238 and U-238, which are provided in Appendix A, Table A-3.

The values assigned for f_s and ϵ/D_e were 0.86 and 40, respectively, based on values provided by Case et al. (1990). Values assigned for U and σ_x were 2.25 m/sec and 2 m, respectively, based on default values provided by the EPA (1987).

References for Section 8

- Case, M. J., S. J. Maheras, M. A. McKenzie-Carter, M. E. Sussman, and P. Voilleque, 1990, *Radioactive Waste Management Complex Performance Assessment*, draft, EG&G Idaho, Inc., EGG-WM-8773, June.
- EPA, 1987, *Interim Protocols for Screening and Follow-up Radon and Radon Decay Product Measurements*, U.S. Environmental Protection Agency, Office of Radiation Programs, EPA 520/1-86-014-1.
- EPA, 1991, *Risk Assessment Guidance for Superfund: Volume 1—Human Health Evaluation Manual (Part B, Development of Risk-based Preliminary Remediation Goals)*, U.S. Environmental Protection Agency, Interim Publication 9285.7-01B, PB92-963333, December.

9. EVALUATION OF PERFORMANCE IN INTRUSION SCENARIOS

9.1 Introduction

Four intrusion scenarios were evaluated:

- Intruder-drilling (acute dose)
- Intruder-excavation (acute dose)
- Intruder-agricultural (chronic dose)
- Intruder-radon (chronic dose).

In the intruder-drilling scenario, a well is drilled directly through the waste. The drilling crew suffers acute exposure to a portion of the waste from external radiation and inhalation by handling the drill cuttings. In the intruder-excavation scenario, a portion of the waste is excavated during construction activities. Again, the onsite workers are acutely exposed. The intruder-agricultural scenario is an extension of the drilling and excavation scenarios. It involves an agricultural pathway and results in chronic exposure. This scenario assumes that a portion of the waste is exhumed through drilling or construction activities (whichever type of activity exhumes the larger amount of waste for the assumed conditions). The waste is then spread and plowed around the site where crops are grown, resulting in chronic exposure by inhalation and direct exposure as well as ingestion of contaminated foodstuffs. In the intruder-radon scenario, an intruder occupies a house with a basement constructed above the waste. The intruder receives an inhalation dose by exposure to radon from the waste that diffuses into the basement.

The specific exposure routes analyzed in these intrusion scenarios are shown in Figure 9-1.

ML5055j27

Figure 9-1. Exposure routes evaluated in the intruder scenarios.

The results of the acute intruder-drilling and acute intruder-excavation scenarios were compared against the 500-mrem acute exposure performance objective in DOE Order 5820.2A. The result of the chronic intruder-agriculture scenario was compared against the 100-mrem/yr continuous exposure performance objective in DOE Order 5820.2A. These scenarios are comparable to the scenarios used by the NRC in 10 CFR 61 to evaluate the land disposal of LLW in commercially operated facilities.

The results of the chronic intruder-radon scenario were compared against two performance objectives. One was the 100-mrem/yr continuous exposure performance objective in DOE Order 5820.2A. The other was the concentration limit of radon in indoor air from EPA (1987).

The effects of radioactive decay and ingrowth were accounted for in all cases. It was also assumed that depletion of the inventory by leaching did not occur, except for the intruder-radon scenario, in which leaching was accounted for. Not considering the effects of leaching produces conservative results. In reality, a portion of the inventory may be leached and transported to a depth where it can no longer be brought to the surface by intruder events.

The radionuclide inventories used in the intruder scenarios are described in Appendices A and B. A homogeneous inventory was used in most cases, wherein it was assumed that the concentrations of radionuclides in the exhumed waste were representative of the average inventory. Additional runs were performed that simulated the effects of separate intrusions into the CH and RH fractions of the refined inventory developed in Appendix A, Section A-9 and of the inventory with additional refinements developed in Appendix A, Section A-10.

To within the level of detail in this study, the intrusion scenarios were assumed to be independent of the disposal location.

The conceptual models, numerical models, assumptions, and parameter values used to calculate doses from the intruder-drilling, intruder-excavation, intruder-agricultural, and intruder-radon scenarios are provided in Sections 9.2, 9.3, 9.4, and 9.5 respectively.

9.2 Intruder-Drilling Scenario

9.2.1 Conceptual Model

The intruder-drilling scenario assumes that an inadvertent intruder drills a water-supply well through the disposal pit and intercepts a column of waste. The intercepted waste is brought to the surface as cuttings. These radioactive cuttings are mixed with (and diluted by) clean cuttings derived from soil that was originally present in the drill hole above the waste. The cuttings were then assumed to be spread in a thin layer over an area surrounding the drill hole and mixed with native surface soil.

The presence of the exhumed waste at the surface results in acute exposure to the drill crew by inhalation of airborne particulates and by direct exposure to penetrating radiation. Predicted doses are affected by the volume of waste excavated, radionuclide concentration, dilution of the waste at the surface, and duration of inhalation and direct exposure. Since decay and ingrowth

are considered in the model, the predicted doses are also a function of the time at which the intrusion occurs.

9.2.2 Numerical Model

Doses from the intruder-drilling scenario were calculated using the short-term environmental dosimetry option of the GENII code (Napier et al. 1988). This numerical model implements the NRC-recommended methodology for exposure pathway assessment and performs the ICRP-recommended dose calculation methodology as described in ICRP 26 and 30. The dose conversion factors used by GENII to calculate dose are based on the most conservative factors contained in DOE 1988a and DOE 1988b. A description of the GENII code is provided in Appendix E.

9.2.3 Assumptions and Parameter Values

This section describes the assumptions and parameter values used in simulating the intruder-drilling scenario. Specific parameter values are provided in Table 9-1. Figure 5-1 shows the layout of the waste disposal pit.

Calculated doses were based on the assumption that a 30-cm-diameter residential water supply well was drilled at the site. Drillers were on site for a total of 40 hr, during which time they were subject to direct exposure from the cuttings, and an additional hour of inhalation exposure, during which time the drill bit intersected the waste.

One supplemental run was performed that extended the 1-hr inhalation exposure time to 8 hr. This supplemental run was performed to examine the possibility that it may take more than 1 hr to drill through a dense ceramic FWF.

Two facility designs were simulated (see Figure 5-1). One design had a 16-ft layer of soil overlying the waste (5 ft of belowgrade backfill plus 11 ft of abovegrade multilayer cover that was conservatively assumed to be all soil). The other design had a 20-ft cover of soil overlying the waste (9 ft of backfill plus the 11-ft cover). The thickness of the soil cover affects intruder-drilling doses by varying the volume of clean soil within the drill hole that is mixed with the waste.

It was further assumed that the exhumed portions of the FWFs are ground totally into respirable particulates by the drilling process. This is a conservative assumption for brittle glass or ceramic FWFs, which tend to fracture into larger pieces. The use of water or mud as a drilling fluid may also significantly reduce the generation of respirable particulates relative to the conservative modeling assumptions. In a limited number of runs that are so identified, particle size distributions that are more realistic were assumed.

The detailed rationale for the intruder-drilling cases evaluated is given in Section 10.4.1.2. The intruder-drilling scenario was conservatively assumed to occur 100 yr after closure of the facility, corresponding to the start of the postinstitutional control period. Previous performance assessments (Maheras et al. 1994) have concluded that the dominant contributors to dose in this type of intruder scenario are nonactinides such as Cs-137. The inventories of these radionuclides during the postinstitutional control period are highest at the beginning of this period.

Table 9-1. Parameter values for intruder-drilling scenario.

Parameter	Value
Inhalation exposure duration while drilling	1 hr
Mass loading factor during drilling process	$1 \times 10^{-4} \text{ g/m}^3$
Intrusion scenario well diameter	30 cm
Length of time an individual is externally exposed to contaminated surface soils	40 hr
Volume of clean surface soil mixed with the drill core of waste and backfill	0.34 m^3 ^a

a. Based on 16-ft cover (5-ft belowgrade backfill plus 11-ft-thick abovegrade engineered cover) and 30-cm-diameter well.

Source: Aaberg and Kennedy 1990.

The multilayer cover has an assumed design life of 500 yr (see Section 5.2). Therefore, it was assumed to be intact 100 yr after closure.

Figure 9-2 illustrates the layout of the disposal pit for the intruder-drilling scenarios 100 yr postclosure.

9.3 Intruder-Excavation Scenario

9.3.1 Conceptual Model

The intruder-excavation scenario assumes that an inadvertent intruder moves onto the site and excavates a basement for a residence. A portion of that excavation disturbs the waste and brings part of it to the surface. The waste brought to the surface is then assumed to be spread in a layer over an area surrounding the excavation site and mixed with native surface soil.

The presence of the exhumed waste at the surface results in acute exposure to the construction crew by inhalation of airborne particulates and by direct exposure to penetrating radiation. Predicted doses are affected by the volume of waste excavated, radionuclide concentration, dilution of the waste at the surface, and duration of inhalation and direct exposure. Since decay and ingrowth are considered in the model, predicted doses are also a function of the time at which intrusion occurs.

9.3.2 Numerical Model

Doses from the intruder-excavation scenario were calculated with the GENII code using the short-term environmental dosimetry option. This model is identical to the one used for the intruder-drilling scenario except for different values of certain input parameters. The GENII model (Napier et al. 1988) implements the NRC-recommended methodology for exposure

Figure 9-2. Disposal pit layout for intruder-drilling scenario, 100 yr postclosure. (At 500 yr postclosure, no credit would be taken for the abovegrade portion of the cover.)

pathway assessment and performs the ICRP-recommended dose calculations as described in ICRP 26 and 30. The dose conversion factors used by GENII to calculate dose are based on the most conservative factors contained in DOE 1988a and DOE 1988b. A description of the GENII code is provided in Appendix E.

9.3.3 Assumptions and Parameter Values

This section describes the assumptions and parameter values used in simulating the intruder-excavation scenario. Specific parameter values are provided in Table 9-2.

The detailed rationale for the intruder-excavation cases evaluated is given in Section 10.4.1.3. The facility has a total of 16 or 20 ft of soil (two cases) overlying the waste. This soil cover is present at facility closure and maintained during the 100-yr period of institutional control. Excavation of an 8-ft-deep basement in this cover material at 100 yr will not disturb the waste in either of the two cases and will result in an insignificant acute dose.

It was assumed that the 11-ft abovegrade cover has a 500-yr lifetime. Excavation of a basement in the cover material at 500 yr is assumed to disturb a layer of waste that is 3 ft thick

Table 9-2. Parameter values for intruder-excavation scenario for cases involving 1-m (3-ft) penetration of the waste.

Parameter	Value
Inhalation and external exposure duration	80 hr (two workweeks)
Mass loading factor for the top layer of soil available for resuspension	$5 \times 10^{-3} \text{ g/m}^3$
Volume of waste and backfill excavated	100 m^3
Volume of clean surface soil mixed with the excavated waste and backfill volume	152 m^3 ^a

a. Based on a 10- × 10-m excavated cross-sectional area and 1 m (3 ft) of waste and backfill around the waste.

Source: Aaberg and Kennedy 1990.

for the shallower burial case (8 ft excavation minus 5 ft backfill equals a 3-ft penetration into the waste). For the deeper burial case, the 8-ft excavation was assumed not to penetrate the top of the waste, which is 9 ft belowgrade. See Figures 9-3 and 9-4, which depict the situation for the two burial depths at 500 yr.

It was assumed in this scenario that the durations of inhalation and external exposure are both equal to 80 hr, which is the equivalent of two work weeks for a maximally exposed member of the construction crew. The mass loading factor used to calculate doses from inhalation was assumed to be $5 \times 10^{-3} \text{ g/m}^3$, which is considerably higher than the value of $1 \times 10^{-4} \text{ g/m}^3$ used in the intruder-drilling scenario. The higher value was used here based on the assumption that excavation equipment generates more respirable fines than does well-drilling equipment.

9.4 Intruder-Agricultural Scenario

9.4.1 Conceptual Model

The intruder-agricultural scenario assumes that waste is brought to the surface by either an intruder-drilling scenario or an intruder-excavation scenario. (Both of these initiating events were included as appropriate for the assumed cover conditions at the time of intrusion.) The waste removed from the well or excavation is then spread and plowed around the site where crops are grown, resulting in chronic exposure by inhalation and direct exposure. In addition, the waste is leached into the root zone, and uptake of nuclides by the crops occurs. The crops are then consumed directly by humans and used to feed livestock, which are then consumed by humans.

The chronic exposure routes include inhalation; direct exposure; and consumption of leafy vegetables, root vegetables, grains, fruit, eggs, milk, beef, pork, and poultry. A maximally exposed

Figure 9-3. Disposal pit layout for intruder-excavation scenario, 500-yr postclosure with 9-ft belowgrade backfill.

Figure 9-4. Disposal pit layout for intruder-excavation scenario, 500-yr postclosure with 5-ft belowgrade backfill.

individual is assumed to reside at the site, consume contaminated foodstuffs, and be subject to the inhalation and direct exposure routes on a near-full-time basis, resulting in chronic rather than acute doses.

9.4.2 Numerical Model

Doses from the intruder-agricultural scenario were calculated using the inhalation, direct exposure, and food chain exposure pathways of the GENII code (Napier et al. 1988). This model is identical to the one used for the intruder-drilling and intruder-excavation scenarios, with the exceptions of the addition of food chain pathways and the calculation of chronic rather than acute doses. The GENII model implements the NRC-recommended methodology for exposure pathway assessment and performs the ICRP-recommended dose calculations as described in ICRP 26 and 30. The dose conversion factors used by GENII to calculate dose are based on the most conservative factors contained in DOE 1988a and DOE 1988b. A description of the GENII code is provided in Appendix E.

9.4.3 Assumptions and Parameter Values

This section describes the assumptions and parameter values used in simulating the intruder-agricultural scenario. Specific parameter values are provided in Table 9-3. Where agricultural and other parameters appear in Table 9-3 that correspond to those for the undisturbed-waste scenario in Table 7-1, the values used were identical.

Two separate types of intruder-agricultural scenarios were evaluated and used as appropriate for the assumed disposal conditions. The first type is initiated by an intruder-drilling event in which drill cuttings are the source of the contaminated material. The second type is initiated by an intruder-excavation event in which the excavated soil/waste mixture is the source of the contaminated material. Parameter values related to these initiating events, such as the volume of waste exhumed, mass loading factors, and the waste-to-soil dilution factors, are identical to the values used in the respective intruder-drilling and intruder-excavation scenarios.

The detailed rationale for the intruder-agricultural cases evaluated is given in Section 10.4.1.4. Intruder-agricultural scenarios were considered for both 16-ft and 20-ft soil cover designs and at both 100 yr and 500 yr postclosure. These times are expected to produce the peak doses for this scenario.

The agricultural exposures for the 100 yr postclosure case were assumed to be initiated by a drilling event through the waste pit (see Figure 9-2) because, for either cover, an excavation event would not penetrate the waste at 100 yr. For the same reason, the agricultural exposures for the case with the 20-ft cover at 500 yr were assumed to be initiated by a drilling event through the waste pit.

For the case with the 16-ft cover at 500 yr, the agricultural exposures were assumed to be initiated by an excavation event, because excavation would bring much more waste to the surface than would drilling. The event would penetrate the waste, as explained in Section 9.3.3 and illustrated in Figure 9-4.

Table 9-3. Parameter values for intruder-agricultural scenario.

Parameter	Value
Weighted-average inhalation exposure duration	6,570 hr/yr
Weighted-average mass loading factor	$1.0 \times 10^{-4} \text{ g/m}^3\text{a}$
Weighted-average external exposure duration	5,825 hr/yr
Volume of waste and backfill mixed with clean surface soil	$0.34 \text{ m}^3\text{a}$
Volume of clean surface soil mixed with contaminated volume of waste and backfill	$2,500 \text{ m}^2 \times 0.15 \text{ m} = 375 \text{ m}^3$
Growing period for hay consumed by milk cows	45 days
Length of growing season for leafy vegetables	45 days
Food holdup time between harvest and consumption of leafy vegetables	1 day
Human consumption rate of leafy vegetables	$11 \text{ kg/yr} \times 0.25^b = 2.8 \text{ kg/yr}$
Length of growing season for root vegetables	90 days
Food holdup time between harvest and consumption of root vegetables	14 days
Human consumption rate of root vegetables	$51 \text{ kg/yr} \times 0.25^b = 12.75 \text{ kg/yr}$
Length of growing season for grains	90 days
Food holdup time between harvest and consumption of grains	14 days
Human consumption rate of grains	$69 \text{ kg/yr} \times 0.25^b = 17.25 \text{ kg/yr}$
Length of growing season for fruit	90 days
Food holdup time between harvest and consumption of fruit	14 days
Human consumption rate of fruit	$46 \text{ kg/yr} \times 0.25^b = 11.5 \text{ kg/yr}$
Leafy-vegetable-yield value	2.0 kg/m^2
Other-vegetable-yield value	4.0 kg/m^2
Eggs-yield value	1.0 kg/m^2
Milk-yield value	1.2 kg/m^2
Beef-yield value	1.2 kg/m^2
Pork-yield value	0.8 kg/m^2
Poultry-yield value	1.0 kg/m^2
Consumption rates of feed or forage by beef cattle	44 kg/day
Consumption rates of feed or forage by milk cows	67 kg/day
Consumption rates of feed or forage by poultry	0.22 kg/day
Consumption rate of drinking water by beef cattle	50 L/day
Consumption rate of drinking water by milk cows	60 L/day
Consumption rate of drinking water by poultry	0.3 L/day
Food holdup time for beef	20 days
Food holdup time for pork	34 days
Food holdup time for poultry	1 day
Food holdup time for milk	1 day

Table 9-3. (continued).

Parameter	Value
Food holdup time for eggs	1 day
Human consumption rate of milk	103 kg/yr
Human consumption rate of beef	59 kg/yr
Human consumption rate of pork	30 kg/yr
Human consumption rate of poultry	9 kg/yr
Human consumption rate of eggs	10 kg/yr

a. Values used for initiation of the intruder-agricultural scenario by a drilling event. If the scenario is initiated by an excavation event, the corresponding values are 5×10^{-3} g/cm³ for the weighted-average mass loading factor and 100 m³ for the volume of waste and backfill mixed with clean surface soil.

b. Based on the results of a U.S. Department of Agriculture survey, only 25% of the vegetables in the exposed individual's diet is derived from a residential garden (Aaberg and Kennedy 1990; Kennedy and Strenge 1992).

Sources: Aaberg and Kennedy 1990; Kennedy and Strenge 1992.

Residents were assumed to reside on site and be subject to inhalation for 6,570 hr/yr (75% of continuous occupancy), and direct exposure for 5,825 hr/yr (66% of continuous occupancy).

Consumption rates of contaminated fruit, grains, and vegetables were based on the assumption that 25% of these categories of food in the diet of the exposed individuals is from a residential garden (Aaberg and Kennedy 1990; Kennedy and Strenge 1992).

9.5 Intruder-Radon Scenario

Section 8 discussed how the transport and chronic inhalation of radon were evaluated for the undisturbed-waste scenario. This section discusses the analogous methodology used for the chronic intruder-radon scenario, which was evaluated in the third phase of the study. The methodology is similar to that used by Maheras et al. (1994).

9.5.1 Conceptual Model

The conceptual model assumes that an inadvertent intruder moves onto the disposal site at least 500 yr postclosure, when the abovegrade portion of the cover has disappeared. The intruder resides in a house with a basement, which is constructed directly over the waste. See Figure 9-5 for the assumed layout of the disposal pit and the house.

Figure 9-5 indicates that, in the case analyzed, the bottom of the 8-ft basement is 1 ft above the top of the waste. This situation corresponds to the deeper burial case in Figure 5-1, with 9 ft of backfill. The more shallow burial case, with only 5 ft of backfill, was considered but was not evaluated for the following reason: In the case with only 5 ft of backfill, the calculated doses for the intruder-agricultural scenario exceed the performance objective by a large margin. The outcome of evaluating the intruder-radon scenario would not alter the conclusion about the

inadequacy of the more shallow burial. The dose from radon would simply add to the dose from the agricultural pathways; the latter dose already exceeds the performance objective. Therefore, only the case with deeper burial was evaluated here.

As in the undisturbed-waste scenario, Rn-222 and its decay products result from the radioactive decay of Pu-238 and U-238. The two FWFs considered in the scenario, PCC and IEB4, release radionuclides by different mechanisms. The radon generated in the PCC is released by diffusion; that generated in the IEB4 is released by dissolution of the FWF. After the radon is released from the FWF, it diffuses through the soil and through the concrete walls of the basement. The radon then circulates through the house and basement, where the indoor intruder receives a dose via the inhalation exposure route.

9.5.2 Numerical Model

The RESRAD computer code (Gilbert et al. 1989), Version 5.19, was used to perform the dose assessments for this scenario.

9.5.3 Assumptions and Parameter Values

Figure 9-5 shows most of the assumptions concerning the disposal unit and the house. The areal dimensions of the house and basement were 30.5 ft by 30.5 ft. RESRAD does not have the capability to model separate radon exposures in the basement and on the main floor of the house, so a total room height of 17.2 ft was used to account for the combined exposure in both areas.

ML5059j27

Figure 9-5. Disposal pit and house layout for intruder-radon scenario, >500 yr postclosure with 9-ft belowgrade backfill.

RESRAD requires a fractional (yr^{-1}) leach rate for each radionuclide. The fractional leach rate for PCC was determined by using the DUST computer code (Sullivan 1993) to calculate a flux out of the disposal facility. By 3,000 yr after the end of institutional control, the flux had decreased to essentially zero ($<10^{-10}$ Ci/cm²-yr). The initial activities of Pu-238 and U-238 were divided by 3,000 yr to give a constant release rate for each radionuclide in Ci/yr. This release was then divided by the initial activity to give the fractional leach rate. For the PCC FWF, the resulting leach rates of the long-lived progenitors of Rn-222 are shown in Table 9-4.

For the IEB4 FWF, the uniform dissolution rate of 5.0×10^{-6} g/cm²-day from Table 6-4 was converted to a fractional leach rate by using the surface area of a waste drum and the concentrations of the radionuclides in the waste. For the IEB4 FWF, the resulting fractional leach rates of the long-lived Rn-222 progenitors are shown in Table 9-4.

The assumed properties of the concrete used to construct the basement are based on data from two instrumented basement structures located at Colorado State University in Fort Collins, Colorado (Gadd 1993).

The data in Konz et al. (1989) were used to estimate that the intruder living in the house would spend 68% of the time at home and indoors.

Table 9-5 lists additional parameters used in evaluating the intruder-radon scenario.

9.5.4 Comparison of Calculational Method for Intruder-Radon Scenario against that for Radon in Undisturbed-Waste Scenario

The method for evaluating the intruder-radon scenario is both similar to and different from the method for evaluating radon transport and exposure in the undisturbed-waste scenario in Section 8. Table 9-6 lists several items for comparison. Where the scenarios were similar, an effort was made to use identical assumptions and parameter values. On many points, however, the differences in the scenarios justified the use of different assumptions and parameter values.

Table 9-4. Fractional leach rates used in the intruder-radon scenario.

Waste form	Pu-238 fractional leach rate (1/yr)	U-238 fractional leach rate (1/yr)	U-234 fractional leach rate (1/yr) ^a	Th-230 fractional leach rate (1/yr) ^a	Ra-226 fractional leach rate (1/yr) ^b
PCC	0.0003	0.0003	0.0003	0.0003	0.0006
IEB4	0.00005	0.00005	0.00005	0.00005	0.0001

a. Leach rate set equal to value for U-238 (Maheras et al. 1994).

b. Leach rate is two times value for U-238 (Maheras et al. 1994).

Table 9-5. Parameters used in intruder-radon scenario.

Parameter	Value	Reference
Thickness of foundation	0.1 m	Gadd (1993)
Bulk density of foundation	2.1 g/cm ³	Gadd (1993)
Porosity of foundation	0.13	Gadd (1993)
Diffusion coefficient of radon in cover	3.0×10^{-6} m ² /s	Gilbert et al. (1989)
Diffusion coefficient of radon in foundation	2.5×10^{-8} m ² /s	Gadd (1993)
Height of room (two levels)	5.25 m	—
Building depth below grade	2.44 m	—
Emanation fraction	0.17	Gadd (1993)

Table 9-6. Similarities and differences in the methods for evaluating radon transport and exposure.

Item	Assumption for radon in undisturbed-waste scenario	Assumption for intruder-radon scenario	Discussion
Location of receptor	Outdoors at downwind edge of disposal unit	Indoors directly above waste	Differences in scenarios
Thickness of cover	5 ft of belowgrade backfill; 11 ft of abovegrade cover for runs <500 yr postclosure; no abovegrade cover thereafter	9 ft of belowgrade backfill; abovegrade cover eroded away	Value in undisturbed-waste scenario selected to give worst-case dose; value in intruder-radon scenario selected because dose from agricultural pathways for shallow-burial case already failed performance objective
Excavation into waste	No excavation	Excavation of basement to depth 1 ft above top of waste	Differences in scenarios
Sources of radon	Pu-238 and U-238 decay chains	Pu-238 and U-238 decay chains	Same in both scenarios
Radon decay chain evaluated	Rn-222 and short-lived decay products generated post-inhalation	Rn-222 and short-lived decay products generated post-inhalation	Same in both scenarios
FWFs evaluated	PCC	PCC, IEB4	Undisturbed-waste scenario assumed no leaching; with this assumption, Rn-222 would exit as a gas from only porous FWFs, so only PCC was evaluated

Table 9-6. (continued).

Item	Assumption for radon in undisturbed-waste scenario	Assumption for intruder-radon scenario	Discussion
Mechanism for release from FWF	Diffusion from PCC	Diffusion from PCC; dissolution of IEB4	Different release mechanisms for different FWFs
Transport through subsurface	Diffusion through soil	Diffusion through soil; diffusion through concrete walls of basement	Different transport pathways in different scenarios
Airborne transport analysis	Simple box model	Model in RESRAD code, which incorporates several airborne transport processes	Different transport pathways in different scenarios
Source term reduction as a result of leaching and groundwater transport	Conservatively neglected	FWF leaching and corresponding inventory reduction, in addition to source zone sorption, as addressed by RESRAD	Intruder-radon method selected to include evaluation of FWF leaching, and to be similar to that in Maheras et al. 1994
Dose model	Dose conversion factors taken from DOE 1988b	Dose conversion factors in RESRAD	None
Times evaluated, yr postclosure	500 yr, to show effect of abovegrade cover; 167,000 yr for maximum dose as determined with DECAYDOS	4600 yr for maximum dose for PCC and 20,000 yr for maximum dose for IEB4, as determined with RESRAD	Factoring in leaching and sorption leads to maximum dose for PCC occurring earlier in intruder-radon scenario than in undisturbed-waste scenario; radon escapes from IEB4 more slowly than from PCC, so maximum dose occurs later

References for Section 9

- Aaberg, R. L., and W. E. Kennedy, 1990, *Definition of Intrusion Scenarios and Example Concentration Ranges for the Disposal of Near-Surface Waste at the Hanford Site*, PNL-6312 UC-702, Pacific Northwest Laboratory.
- DOE, 1988a, *External Dose-Rate Conversion Factors for Calculation of Dose to the Public*, DOE/EH-0070, U.S. Department of Energy, March.
- DOE, 1988b, *Internal Dose-Rate Conversion Factors for Calculation of Dose to the Public*, DOE/EH-0071, U.S. Department of Energy, July.
- EPA, 1987, *Interim Protocols for Screening and Follow-up Radon and Radon Decay Product Measurements*, Office of Radiation Programs, EPA 520/1-86-014-1, U.S. Environmental Protection Agency.
- Gadd, M. S., 1993, *The Origins and Pathways of ^{222}Rn Entering into Basement Structures*, Ph.D. Dissertation, Colorado State University, Fort Collins, Colorado.
- Gilbert, T. L., C. Yu, Y. C. Yuan, A. J. Zielen, M. J. Jusko, and A. Wallo III, 1989, *A Manual for Implementing Residual Radioactive Material Guidelines*, ANL/ES-160, DOE/CH/8901.
- Kennedy, W. E., and D. L. Streng, 1992, *Residual Radioactive Contamination from Decommissioning: Technical Basis for Translating Contamination Levels to Annual Total Effective Dose Equivalent*, NUREG/CR-5512, Pacific Northwest Laboratory.
- Konz, J. J., K. Lisi, E. Friebele, and D. A. Dixon, 1989, *Exposure Factors Handbook*, EPA/600/8-89/043, U.S. Environmental Protection Agency, July.
- Maheras, S. J., A. S. Rood, S. O. Magnuson, M. E. Sussman, and R. N. Bhatt, 1994, *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment*, EG&G Idaho, Inc., EGG-WM-8773, May.
- Napier, B. A., R. A. Peloquin, D. L. Streng, and J. V. Ramsdell, 1988, *GENII—The Hanford Environmental Radiation Dosimetry Software System: Conceptual Representation*, PNL-6584, Vol. 1, Pacific Northwest Laboratory.
- Sullivan, T. M., 1993, *DUST Disposal Unit Source Term: Data Input Guide*, NUREG/CR-6041, BNL-NUREG-52375, Brookhaven National Laboratory, April.

10. RESULTS

10.1 Introduction and Summary

This section provides the results of the evaluations, which covered a range of assumptions and parameter values. This preliminary parametric PA evaluated briefly the effects of several factors on predicted doses to members of the public and inadvertent intruders. The study is not a detailed evaluation of one specific set of conditions.

The effects on the dose as a result of many factors were investigated in various combinations. Not all combinations were considered, but the ones that were selected show most efficiently the effects of these factors on the results. The approach was basically to perform a group of runs (Runs 1 through 4, 6, and 8 through 19) using an original set of assumptions and parameter values, then to investigate the effects of changes in the assumptions and parameter values, taken one, two, or three at a time.

Table 10-1 lists the performance objectives and the top-level PA results. As expected, compliance with each performance objective varies from one FWF to another and from one disposal site to another.

The results for the undisturbed-waste scenario—groundwater pathway are presented in Section 10.2. Because a large number of runs were performed, the results of each run are compiled in Table 10-2 for convenience. The purpose of each run is given under the description in the last column. Detailed information regarding each run is provided in the section whose number is listed in the next-to-last column of the table.

The results for the undisturbed-waste scenario—atmospheric pathway are presented in Section 10.3.

The results for the intrusion scenarios are presented in Section 10.4. Again, because a large number of runs were performed, the results of each run are compiled in Table 10-3 for convenience. The results of runs to confirm the assumed time of peak intrusion dose, however, are not included in the table.

As stated previously, to a first approximation, the results of the intrusion scenarios are independent of the disposal site. Therefore, no site dependence of the results is indicated in Table 10-3.

Because the calculational scheme for the intruder-radon scenario differed from that for the other intrusion scenarios, the results are given in Section 10.4.12, rather than in Table 10-3.

Section 10.5 provides a qualitative prediction of the effects of secondary FWFs on the performance of the disposal system.

Table 10-1. Performance objectives and top-level results.

Regulation	Exposure group	Performance objectives limit	Top-level results
<u>DOE Orders</u>			
5820.2A, Ch. III	Public	25 mrem/yr EDE (all pathways but air)	For Sites B and C, all FWFs pass. For Site A, all FWFs pass except for PCC and untreated waste under very conservative assumptions. For Site D, all FWFs fail. For Site E, all FWFs except IEB4 fail.
	Inadvertent intruder	100 mrem/yr EDE (chronic exposure)	With 9-ft backfill plus 11-ft cover, all FWFs pass if refined assumptions are made on particle size distribution or on nonactinide inventory for IEB and IEB4 FWFs. With 5-ft backfill plus 11-ft cover, all FWFs fail intruder-agricultural scenario.
		500 mrem EDE (acute exposure)	For intruder-drilling and intruder-excavation scenarios, all FWFs pass except for RH portion.
<u>EPA</u>			
40 CFR 61, Subpart H	Public	10 mrem/yr EDE (air emissions)	Dose expected to be much less than limit in all cases, based on results in Maheras et al. (1994).
40 CFR 141	Public	4 mrem/yr from beta emitters and photon emitters	For Sites A, B, and C, all FWFs pass. For Sites D and E, all FWFs except IEB4 fail.
		15 pCi/L total alpha emitters (including Ra-226, excluding U and Rn)	For Sites A, B, and C, all FWFs pass. For Sites D and E, all FWFs fail.
		5 pCi/L combined Ra-226 and Ra-228	All FWFs pass at all sites.
		30 pCi/L U	For Sites A, B, and C, all FWFs pass. for Sites D and E, all FWFs except IEB4 fail.
EPA (1987)	Inadvertent intruder	4 pCi/L Rn concentration in indoor air	All FWFs pass for intruder-radon scenario. Limit is not applicable for undisturbed-waste scenario because intruder is not indoors.

Table 10-2. Compilation of results for the undisturbed-waste scenario—groundwater transport pathway.

Run number	FWF	Site	Maximum dose (mrem/yr) limit = 25 mrem/yr ^a	Time of maximum dose (yr)	Dominant decay-chain contributing to maximum dose	Section where discussed	Description/purpose
1	PCC	A	3.1	2,900,000	Am-241	10.2.2	Least favorable FWF at less favorable disposal site.
2A	PCC	C	2.3×10^{-3}	43,000,000	Am-241	10.2.2	Least favorable FWF at most favorable disposal site. (Runs 2A, 2B, and 2C reflect three assumed groundwater velocities.)
2B	PCC	C	4.7×10^{-4}	44,000,000	Am-241	10.2.2	
2C	PCC	C	1.9×10^{-6}	57,000,000	Am-241	10.2.2	
3	IEB4	A	0.011	2,900,000	Am-241	10.2.2	Most favorable FWF at less favorable disposal site.
4A	IEB4	C	3.5×10^{-4}	43,000,000	Am-241	10.2.2	Most favorable FWF at most favorable disposal site. (Runs 4A, 4B, and 4C reflect three different assumed groundwater velocities.)
4B	IEB4	C	7.2×10^{-5}	44,000,000	Am-241	10.2.2	
4C	IEB4	C	2.8×10^{-7}	57,000,000	Am-241	10.2.2	
5	IEB	B	0.0015	7,000,000	Am-241	10.2.3	This set of runs evaluates effect of TRU radionuclide concentration in FWF.
6	IEB	B	0.015	7,000,000	Am-241	10.2.1	
7	IEB	B	0.15	7,000,000	Am-241	10.2.3	
8	IEB4	B	0.0024	7,000,000	Am-241	10.2.1	This set of runs (along with Run 6) evaluates different FWFs at given disposal site.
9	PCC	B	0.78	7,000,000	Am-241	10.2.1	
10	FUETAP	B	0.52	7,000,000	Am-241	10.2.1	
11	BSG	B	0.080	7,000,000	Am-241	10.2.1	
12	ASG	B	0.015	7,000,000	Am-241	10.2.1	

Table 10-2. (continued).

Run number	FWF	Site	Maximum dose (mrem/yr) limit = 25 mrem/yr ^a	Time of maximum dose (yr)	Dominant decay-chain contributing to maximum dose	Section where discussed	Description/purpose
13	PCC	D	1,100 ^b	9,200	U-233	10.2.2	This set of runs (along with Runs 1, 2, 3, 4, 8, and 9) evaluates different disposal sites.
14	PCC	E	3,400 ^b	470	U-233	10.2.2	
15	IEB4	D	44 ^b	16,000	U-233	10.2.2	
16	IEB4	E	20	10,000	U-233	10.2.2	
17	ASG	A	0.066	2,900,000	Am-241	10.2.11	Most favorable glass at less favorable disposal site.
18	IEB	A	0.067	2,900,000	Am-241	10.2.11	Least favorable ceramic at less favorable disposal site.
17RLR	ASG	A	0.12	2,900,000	Am-241	10.2.11	Same as Run 17, except that revised leach rates are used.
18RLR	IEB	A	0.083	2,900,000	Am-241	10.2.11	Same as Run 18, except that revised leach rates are used.
3RLR	IEB4	A	0.023	2,900,000	Am-241	10.2.11	Same as Run 3, except that revised leach rates are used.
12RLR	ASG	B	0.027	7,000,000	Am-241	10.2.11	Same as Run 12, except that revised leach rates are used.
6RLR	IEB	B	0.02	7,000,000	Am-241	10.2.11	Same as Run 6, except that revised leach rates are used.
8RLR	IEB4	B	0.0054	7,000,000	Am-241	10.2.11	Same as Run 8, except that revised leach rates are used.
18TDLR	IEB	A	2.1×10^{-3}	2,900,000	Am-241	10.2.12	Same as Run 18, except that time-dependent leach rates are used.
6TDLR	IEB	B	4.7×10^{-4}	7,000,000	Am-241	10.2.12	Same as Run 6, except that time-dependent leach rates are used.
1K1	PCC	A	12	950,000	Am-241	10.2.8	Same as Run 1, except that K_d values are lowered by factor of 3.
1K2	PCC	A	28 ^b	290,000	Am-241	10.2.8	Same as Run 1, except that K_d values are lowered by factor of 10.
9K1	PCC	B	7.4	2,300,000	Am-241	10.2.8	Same as Run 9, except that K_d values are lowered by factor of 3.
9K2	PCC	B	24	700,000	Am-241	10.2.8	Same as Run 9, except that K_d values are lowered by factor of 10.

Table 10-2. (continued).

Run number	FWF	Site	Maximum dose (mrem/yr) limit = 25 mrem/yr ^a	Time of maximum dose (yr)	Dominant decay-chain contributing to maximum dose	Section where discussed	Description/purpose
13K1	PCC	D	1,800 ^b	8,300	Pu-239	10.2.8	Same as Run 13, except that K_d values are lowered by factor of 3.
13K2	PCC	D	6,800 ^b	2,900	Pu-239	10.2.8	Same as Run 13, except that K_d values are lowered by factor of 10.
14K1	PCC	E	5,400 ^b	880,000	Am-241	10.2.8	Same as Run 14, except that K_d values are lowered by factor of 3.
14K2	PCC	E	14,000 ^b	270,000	Am-241	10.2.8	Same as Run 14, except that K_d values are lowered by factor of 10.
6GC1	IEB	B	0.052	7,000,000	Am-241	10.2.4	Same as Run 6, except that FWF surface area is increased by factor of 10.
6GC2	IEB	B	0.29	7,000,000	Am-241	10.2.4	Same as Run 6, except that FWF surface area is increased by factor of 100.
6GC3	IEB	B	0.65	7,000,000	Am-241	10.2.4	Same as Run 6, except that FWF surface area is increased by factor of 1,000.
6GC4	IEB	B	1.2	7,000,000	Am-241	10.2.4	Same as Run 6, except that FWF surface area is increased by factor of 10,000.
6GC5	IEB	B	1.2	7,000,000	Am-241	10.2.4	Same as Run 6, except that FWF surface area is increased by factor of 100,000.
9C1	PCC	B	1.1	7,000,000	Am-241	10.2.4	Same as Run 9, except that FWF diffusion path length is decreased by factor of 10.
9C2	PCC	B	1.1	7,000,000	Am-241	10.2.4	Same as Run 9, except that FWF diffusion path length is decreased by factor of 100.
1IR1	PCC	A	6.1	760,000	Am-241	10.2.5	Same as Run 1, except that infiltration rate of 5 cm/yr is used.
1IR2	PCC	A	6.8	380,000	Am-241	10.2.5	Same as Run 1, except that infiltration rate of 10 cm/yr is used.
9IR1	PCC	B	4.4	1,700,000	Am-241	10.2.5	Same as Run 9, except that infiltration rate of 5 cm/yr is used.
9IR2	PCC	B	5.7	840,000	Am-241	10.2.5	Same as Run 9, except that infiltration rate of 10 cm/yr is used.

Table 10-2. (continued).

Run number	FWF	Site	Maximum dose (mrem/yr) limit = 25 mrem/yr ^a	Time of maximum dose (yr)	Dominant decay-chain contributing to maximum dose	Section where discussed	Description/purpose
13IR2	PCC	D	970 ^b	7,500	U-233	10.2.5	Same as Run 13, except that infiltration rate of 80 cm/yr is used.
13IR2	PCC	D	920 ^b	6,900	U-233	10.2.5	Same as Run 13, except that infiltration rate of 120 cm/yr is used.
14IR1	PCC	E	2,900 ^b	270,000	Am-241	10.2.5	Same as Run 14, except that infiltration rate of 5 cm/yr is used.
14IR2	PCC	E	3,000 ^b	130,000	Am-241	10.2.5	Same as Run 14, except that infiltration rate of 10 cm/yr is used.
1LDW	PCC	A	0.0012	3,400,000	Am-241	10.2.9	Same as Run 1, except that receptor well is 215 km downgradient.
1UTR	Untreated	A	3.6	2,900,000	Am-241	10.2.10	Same as Run 1, except that waste is untreated.
9UTR	Untreated	B	0.92	7,000,000	Am-241	10.2.10	Same as Run 9, except that waste is untreated.
13UTR	Untreated	D	780 ^b	9,500	U-233	10.2.10	Same as Run 13, except that waste is untreated.
14UTR	Untreated	E	7,000 ^b	470	U-233	10.2.10	Same as Run 14, except that waste is untreated.
1NB	PCC	A	6.2	720,000	Am-241	10.2.6	Same as Run 1, except that no credit is taken for vadose zone basalt travel time.
1IR2NB	PCC	A	7.7	140,000	U-233	10.2.6	Same as Run 1IR2, except that no credit is taken for vadose zone basalt travel time.
1SZS	PCC	A	0.23	2,900,000	Am-241	10.2.7	Same as Run 1, except that source-zone sorption is used.
9SZS	PCC	B	5.5×10^{-2}	7,000,000	Am-241	10.2.7	Same as Run 9, except that source-zone sorption is used.
13SZS	PCC	D	1,100 ^b	9,200	U-233	10.2.7	Same as Run 13, except that source-zone sorption is used.
14SZS	PCC	E	3,400 ^b	470	U-233	10.2.7	Same as Run 14, except that source-zone sorption is used.
1IR2NBSZS	PCC	A	2.9	140,000	U-233	10.2.7	Same as Run 1IR2NB, except that source-zone sorption is used.
1TBF	PCC	A	2.3×10^{-2}	2,900,000	Am-241	10.2.7	Same as Run 1, except that tailored backfill is assumed.

Table 10-2. (continued).

Run number	FWF	Site	Maximum dose (mrem/yr) limit = 25 mrem/yr ^a	Time of maximum dose (yr)	Dominant decay-chain contributing to maximum dose	Section where discussed	Description/purpose
9TBF	PCC	B	4.8×10^{-3}	7,000,000	Am-241	10.2.7	Same as Run 9, except that tailored backfill is assumed.
18TBF	IEB	A	3.7×10^{-5}	2,900,000	Am-241	10.2.7	Same as Run 18, except that tailored backfill is assumed.
6TBF	IEB	B	6.6×10^{-6}	7,000,000	Am-241	10.2.7	Same as Run 6, except that tailored backfill is assumed.
11R2NBTBF	PCC	A	0.25	840,000	U-233	10.2.7	Same as Run 11R2NB, except that tailored backfill is assumed.
11R2NBUTR	Untreated	A	8.5	140,000	U-233	10.2.10	Same as Run 11R2NB, except that waste is untreated.
1UTRTORT	Untreated	A	3.7	2,900,000	Am-241	10.2.10	Same as Run 1UTR, except that higher diffusion coefficient for sludge is used.
1UTRDIFF	Untreated	A	2.8	2,900,000	Am-241	10.2.10	Same as Run 1UTR, except that radionuclides are assumed to be released by diffusion mechanism.
1UTRSUR	Untreated	A	3.7	2,900,000	Am-241	10.2.10	Same as Run 1UTR, except that radionuclides are assumed to be released by surface washoff mechanism.
1UTH	PCC	A	3.1	2,900,000	Am-241	10.2.16	Same as Run 1, except that U-233 and Th-232 inventories are increased.
3UTH	IEB4	A	0.011	2,900,000	Am-241	10.2.16	Same as Run 3, except that U-233 and Th-232 inventories are increased.
1INT	PCC	A	0.025	2,900,000	Am-241	10.2.13	Same as Run 1, except that chemical interactions between radionuclides and PCC FWF are included.
9INT	PCC	B	0.0066	7,000,000	Am-241	10.2.13	Same as Run 9 except that chemical interactions between radionuclides and PCC FWF are included.
1IAR	PCC	A	3.1	2,900,000	Am-241	10.2.15	Same as Run 1, except that inventory with addition refinements is used.

Table 10-2. (continued).

Run number	FWF	Site	Maximum dose (mrem/yr) limit = 25 mrem/yr ^a	Time of maximum dose (yr)	Dominant decay-chain contributing to maximum dose	Section where discussed	Description/purpose
13IAR	PCC	D	1100 ^b	9,200	U-233	10.2.15	Same as Run 13, except that inventory with addition refinements is used.
14IAR	PCC	E	3400 ^b	470	U-233	10.2.15	Same as Run 14, except that inventory with addition refinements is used.
4AIAR	IEB4	C	0.75	3,200	C-14	10.2.15	Same as Run 4A, except that inventory with addition refinements is used.
1UTRIAR	Untreated	A	3.7	2,900,000	Am-241	10.2.15	Same as Run 1UTR, except that inventory with addition refinements is used.
1VTF	PCC	A	2.9	2,900,000	Am-241	10.2.17	Same as Run 1, except that vault type disposal facility is assumed.
9VTF	PCC	B	0.72	7,000,000	Am-241	10.2.17	Same as Run 9, except that vault type disposal facility is assumed.
14VTF	PCC	E	2100 ^b	470	U-233	10.2.17	Same as Run 14, except that vault type disposal facility is assumed.
18VTF	IEB	A	0.068	2,900,000	Am-241	10.2.17	Same as Run 18, except that vault type disposal facility is assumed.
6VTF	IEB	B	0.017	7,000,000	Am-241	10.2.17	Same as Run 6, except that vault type disposal facility is assumed.
19	IEB	E	150 ^b	22,000	U-233	10.2.17	IEB FWF at Site E.
19VTF	IEB	E	150 ^b	22,000	U-233	10.2.17	Same as Run 19, except that vault type disposal facility is assumed.

a. Source: DOE Order 5820.2A.

b. EDE exceeds the limit set forth in DOE Order 5820.2A.

Table 10-3. Compilation of results for the intrusion scenarios.

Final waste form	Inventory ^a	Special conditions	Intruder-drilling scenario 16-ft soil cover intrusion occurs at 100 yr (mrem) limit = 500 mrem ^b	Intruder-drilling scenario 20-ft soil cover intrusion occurs at 100 yr (mrem) limit = 500 mrem ^b	Intruder-excavation scenario 16-ft soil cover intrusion occurs at 500 yr (mrem) limit = 500 mrem ^b	Intruder-agricultural scenario 16-ft soil cover intrusion occurs at 500 yr (mrem) limit = 100 mrem/yr ^b	Intruder-agricultural scenario 20-ft soil cover intrusion occurs at 100 yr (mrem) limit = 100 mrem/yr ^b	Intruder-agricultural scenario 20-ft soil cover intrusion occurs at 500 yr (mrem) limit = 100 mrem/yr ^b
PCC	Simplified	NA	23	18	96	1,400 ^c	46	NE
FUETAP	Simplified	NA	23	18	96	1,400 ^c	46	NE
10-9	IEB	Simplified	49	42	220	3,100 ^c	110 ^c	22
		Simplified with 10 nCi/g TRU	45	39	22	310 ^c	47	NE
		Simplified with 1000 nCi/g TRU	84	72	2,200 ^c	31,000 ^c	690 ^c	NE
	IEB4	Simplified	59	49	260	3,600 ^c	120 ^c	25
	BSG	Simplified	23	18	96	1,400 ^c	46	NE
ASG	Simplified	NA	46	36	190	2,700 ^c	92	NE
UTR	Refined-Combined	NA	12	9.6	58	820 ^c	24	NE
PCC	Refined-RH	NA	13,000 ^c	NE	92	1,300 ^c	NE	NE
PCC	Refined-CH	NA	2.7	NE	96	1,300 ^c	NE	NE
PCC	Refined-Combined	NA	19	NE	96	1,300 ^c	NE	NE
IEB	Refined-RH	NA	29,000 ^c	NE	210	3,100 ^c	NE	NE
IEB	Refined-CH	NA	6.2	NE	220	3,000 ^c	NE	NE
IEB	Refined-combined	NA	43	NE	220	3,000 ^c	97	22
IEB4	Refined-combined	NA	NE	NE	NE	NE	110 ^c	25

Table 10-3. (continued).

Final waste form	Inventory ^a	Special conditions	Intruder-drilling scenario	Intruder-drilling scenario	Intruder-excavation scenario	Intruder-agricultural scenario	Intruder-agricultural scenario	Intruder-agricultural scenario	
			16-ft soil cover intrusion occurs at 100 yr (mrem) limit = 500 mrem ^b	20-ft soil cover intrusion occurs at 100 yr (mrem) limit = 500 mrem ^b	16-ft soil cover intrusion occurs at 500 yr (mrem) limit = 500 mrem ^b	16-ft soil cover intrusion occurs at 500 yr (mrem) limit = 100 mrem/yr ^b	20-ft soil cover intrusion occurs at 100 yr (mrem) limit = 100 mrem/yr ^b	20-ft soil cover intrusion occurs at 500 yr (mrem) limit = 100 mrem/yr ^b	
PCC	Additional refinements -RH	NA	2,000 ^c	1,700 ^c	96	1,400 ^c	1,700 ^c	10	
PCC	Additional refinements -CH	NA	3	2.5	96	1,300 ^c	29	9.9	
PCC	Additional refinements -combined	NA	5.4	3.6	96	1,300 ^c	31	9.9	
10-10	IEB	Additional refinements -RH	NA	4,500 ^c	3,600 ^c	220	3,100 ^c	3,800 ^c	23
	IEB	Additional refinements -CH	NA	6.7	5.7	220	3,000 ^c	67	22
	IEB	Additional refinements -combined	NA	15	12	220	3,100 ^c	75	22
	IEB4	Additional refinements -RH	NA	5,500 ^c	4,500 ^c	260	3,600 ^c	4,400 ^c	26
	IEB4	Additional refinements -CH	NA	8.4	6.6	260	3,600 ^c	78	26
	IEB4	Additional refinements -combined	NA	21	17	260	3,600 ^c	83	26
	IEB	Simplified	Increased exposure time during drilling	NE	44	NE	NE	NE	NE

Table 10-3. (continued).

Final waste form	Inventory ^a	Special conditions	Intruder-drilling scenario 16-ft soil cover intrusion occurs at 100 yr (mrem) limit = 500 mrem ^b	Intruder-drilling scenario 20-ft soil cover intrusion occurs at 100 yr (mrem) limit = 500 mrem ^b	Intruder-excavation scenario 16-ft soil cover intrusion occurs at 500 yr (mrem) limit = 500 mrem ^b	Intruder-agricultural scenario 16-ft soil cover intrusion occurs at 500 yr (mrem) limit = 100 mrem/yr ^b	Intruder-agricultural scenario 20-ft soil cover intrusion occurs at 100 yr (mrem) limit = 100 mrem/yr ^b	Intruder-agricultural scenario 20-ft soil cover intrusion occurs at 500 yr (mrem) limit = 100 mrem/yr ^b
IEB	Simplified	Refined particle-size distribution	NE	42	NE	NE	30	NE
ASG	Simplified	Refined particle-size distribution	NE	36	NE	NE	26	NE
IEB4	Simplified	Refined particle-size distribution	NE	49	NE	NE	37	NE
BSG	Simplified	Refined particle-size distribution	NE	18	NE	NE	14	NE
PCC or FUETAP	Simplified	Refined particle-size distribution	NE	18	NE	NE	13	NE

Note: Intruder runs are not site-specific.

a. Unless otherwise indicated, all inventories had a TRU concentration of 100 nCi/g.

b. Source: DOE Order 5820.2A.

c. EDE exceeds the limit set forth in DOE Order 5820.2A.

NA = Not applicable.

NE = Not evaluated.

10.2 Results for Undisturbed-Waste Scenario: Groundwater Pathway

This section discusses the results of simulating exposure from the groundwater pathway for the undisturbed-waste scenario. The predicted performance of the various disposal systems is discussed in terms of the effects of key parameters on the predicted maximum annual EDEs.

As discussed in Section 4.2.1, performance was evaluated based on the maximum EDE to a receptor withdrawing groundwater from a well drilled 100 m (328 ft) from the downgradient edge of the disposal facility. The one exception was for the analysis of the effect of distance to the receptor well, which was evaluated on the maximum EDE at a receptor well 215 km (134 mi) from the downgradient edge of the disposal facility.

No time limit was imposed on the occurrence of the maximum EDE.

10.2.1 Effects of FWF on Results

In the initial runs, all six treated FWFs were evaluated at Site B. Table 10-4 shows the results of the six runs.

In all cases, the predicted maximum EDE is well below the regulatory limit of 25 mrem/yr. Differences in maximum EDEs are the result primarily of differences in radionuclide release rates from the FWFs. A higher dissolution rate (or higher diffusion constants in the case of PCC and FUETAP) causes the FWF to release its inventory over a shorter time period, which results in higher doses. The time of maximum dose was delayed only about 4,000 yr (out of about 7,000,000 yr) for the FWF having the best immobilization characteristics, compared with the FWF having the poorest immobilization characteristics.

An additional parameter that affects dose is the radionuclide availability. All waste forms were assumed to have a 100% availability, except for IEB4, which has a 30% availability for the actinides and Sr-90. This reduced availability contributes to lower doses from IEB4.

Table 10-4. Effects of FWF on performance at Site B.

Run number	FWF	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
9	PCC	0.78	7.0E+06
10	FUETAP	0.52	7.0E+06
11	BSG	0.080	7.0E+06
12	ASG	0.015	7.0E+06
6	IEB	0.015	7.0E+06
8	IEB4	0.0024	7.0E+06

10.2.2 Effects of Disposal Site on Results

The effect of the choice of disposal site on the results was evaluated primarily in two sets of runs. Runs 1, 9, 2, 13, and 14 evaluated the performance of PCC at Sites A, B, C, D, and E, respectively. Runs 3, 8, 4, 15, and 16 evaluated the performance of IEB4 at Sites A, B, C, D, and E, respectively. Of all the treated FWFs evaluated, IEB4 has the best immobilization characteristics for groundwater transport, while PCC has the poorest.

For both FWFs, Sites A, B, and C provided acceptable performance with respect to the 25-mrem/yr EDE limit. At Site D, both FWFs exceed the 25-mrem/yr limit. At Site E, PCC exceeds the limit and IEB4 is just barely within the limit. The results are shown in Table 10-5 and discussed below.

Sites A and B. The results show that, for these preliminary assumptions and parameter values, both Sites A and B are acceptable with respect to the 25-mrem/yr standard. Site B offers the better performance of the two. The differences in site performance are attributed to the site hydrologic properties. Site B doses are lower than those from Site A due to the presence of 80 m of sediment within the vadose zone at Site B. Although the vadose zone at Site A consists of 167 m of fractured basalt plus 9 m of sediment, the 80 m of sediment at Site B provides a significantly greater degree of retardation. This results in a radionuclide travel time about 2.4 times that at Site A. This increased travel time allows for more radionuclide decay before reaching the water table.

Table 10-5. Effects of disposal site on performance.

Run number	FWF	Site	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
1	PCC	A	3.1	2.9E+06
9	PCC	B	0.78	7.0E+06
2	PCC	C	2.3 x 10E-03 to 1.9 x 10E-06 ^a	4.3E+07 to 5.7E+07
13	PCC	D	1,100	9.2E+03
14	PCC	E	3,400	4.7E+02
3	IEB4	A	0.011	2.9E+06
8	IEB4	B	0.0024	7.0E+06
4	IEB4	C	3.5 x 10E-04 to 2.8 x 10E-07 ^a	4.3E+07 to 5.7E+07
15	IEB4	D	44	1.6E+04
16	IEB4	E	20	1.0E+04

a. Range of doses reflects impact of range of very small values assumed for the near-zero groundwater velocity.

Site C. Prediction of doses at Site C via this pathway is complicated by uncertainties concerning the groundwater velocity within the saturated zone. Saturated-zone velocities are reported to be near zero or below the limits of measurement. Two sets of runs were performed to assess the effects of varying the saturated-zone groundwater velocity on dose. The first set, consisting of Runs 2A, 2B, and 2C, considered the disposal of PCC at Site C. The other set, consisting of Runs 4A, 4B, and 4C, considered the disposal of IEB4 at Site C. Groundwater velocities of 1.0, 0.1, and 0.01 m/yr (asymptotically approaching zero) were used to assess the impact on the dose. The results are shown in Table 10-6.

The doses at Site C are between 4 and 8 orders of magnitude below the 25-mrem/yr limit, indicating that Site C offers the best performance of the sites evaluated. The very long travel times are a result primarily of the travel time through the vadose zone.

Flow within the saturated zone contributes an additional increment of travel time, depending upon the choice of groundwater velocity. The transport model for the saturated zone considers the combined effects of advection, adsorption, dispersion, and diffusion. When groundwater velocities are extremely small, diffusion dominates transport within the saturated zone. The doses and travel times are similar for the least favorable FWF (PCC) and the most favorable FWF (IEB4), indicating that the doses at Site C are dominated by flow-path processes rather than by leaching processes.

Site D. Site D has a high infiltration rate and a relatively thin vadose zone. These characteristics combine to give Site D a relatively short groundwater travel time. In addition, the K_d for uranium at Site D is lower than at Sites A, B, and C.

The maximum dose for Site D is from the U-233 decay chain and, as shown in Table 10-5, occurs at 9,200 yr for PCC and 16,000 yr for IEB4. For Sites A and B, the maximum dose is from the Am-241 decay chain and occurs at 2.9 and 7 million yr, respectively, for either PCC or IEB4. This difference in travel time for uranium is the reason Site D exceeds the performance objective.

Table 10-6. Effects of FWF and saturated-zone groundwater velocity on performance at Site C.

Run number	FWF	Groundwater velocity (m/yr)	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
2A	PCC	1.0	2.3×10^{-3}	4.3E+07
2B	PCC	0.1	4.7×10^{-4}	4.4E+07
2C	PCC	0.01	1.9×10^{-6}	5.7E+07
4A	IEB4	1.0	3.5×10^{-4}	4.3E+07
4B	IEB4	0.1	7.2×10^{-5}	4.4E+07
4C	IEB4	0.01	2.8×10^{-7}	5.7E+07

Site E. Site E is a semiarid site with travel time and infiltration rate similar to Sites A and B, except that the K_d values at Site E are much smaller than at the other sites. The K_d of U-233 is zero, so it takes only about 450 yr to reach the well. This short radionuclide travel time allows very little decay before the U-233 reaches the well and is the reason for the high dose at Site E.

Reasons for Differences. Some of the performance comparisons between the sites studied were presented in this section. Additional site-specific effects are evident in several other parts of Section 10. The reasons for the differences, which are often quite large if comparing doses at Sites A and B against corresponding doses at Sites D and E, are discussed in more detail in Section 10.2.18. In addition, the caution stated in Section 6.1, about the use of generic models for predictions of site-specific performance, is reiterated.

10.2.3 Effects of TRU Radionuclide Activity Concentration on Results

The effects of the concentration of TRU radionuclide activity in FWFs were evaluated in Runs 5, 6, and 7. These runs assumed an IEB FWF disposed of at Site B. All parameters were held constant in these three runs except for the TRU activity in the IEB, which was set at 10, 100, and 1,000 nCi/g, respectively.

The case with 100 nCi/g corresponds to the actinide inventory used throughout this study. This concentration of TRU radionuclides is the maximum possible for INEL ALLW. The concentration of 10 nCi/g is the minimum possible for INEL ALLW. Waste with a 1,000 nCi/g TRU concentration would be managed as TRU waste, not ALLW. This concentration was used only for the purpose of the parametric evaluations.

Results are shown in Table 10-7.

The predicted peak annual EDEs are 1.5E-03, 1.5E-02, and 1.5E-01 mrem/yr, corresponding to TRU concentrations of 10, 100, and 1,000 nCi/g. In all three cases, the peak dose was from the Am-241 decay chain and occurred at 7.0E+06 yr. This trend in EDEs is reasonable, since a dose from a given radionuclide should be linearly proportional to the activity concentration of that radionuclide in the FWF, as long as all other parameters are held constant. The arrival times were constant because the IEB dissolution rate was held constant for all three TRU concentrations.

Table 10-7. Effects of TRU activity concentration on performance of IEB at Site B.

Run number	TRU activity (nCi/g)	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
5	10	0.0015	7.0E+06
6	100	0.015	7.0E+06
7	1,000	0.15	7.0E+06

10.2.4 Effects of FWF Degradation on Results

Except as indicated otherwise, all of the results presented in this report are based on the assumption that the radionuclide release rate from each FWF is constant until the inventory is depleted. At that time the release rate goes to zero. In reality, the release rate changes with time as a result of two factors. One factor is the change in the diffusion coefficients (in the case of cements) or the dissolution rate (in the case of ceramics and glasses). This factor relates to the complex changes in chemical processes occurring over time. For ceramics and glasses, this factor is addressed in Section 10.2.12.

The second factor relates to the physical degradation of the FWF over time. This factor leads to a reduced length of the diffusion path (in the case of the cements) or an increased surface area (in the case of the ceramics and glasses). This factor was investigated here.

Two sets of evaluations were performed to simulate the performance of degraded FWFs at Site B. The first set consisted of Runs 9, 9C1, and 9C2. Run 9 assumed a monolithic PCC FWF in the shape of a 55-gal drum. Run 9C1 assumed that the original shape was broken into pieces so that the diffusion path length was reduced by a factor of 10. In Run 9C2, the diffusion path length was reduced by an additional factor of 10. The diffusion-limited release model discussed in Section 6.2.3.3 was used in these three runs, each of which had a different average length of the diffusion path, corresponding to the size of the pieces of the FWF.

The second set of analyses consisted of Runs 6, 6GC1, 6GC2, 6GC3, 6GC4, and 6GC5. Run 6 assumed a monolithic IEB FWF in the shape of a 55-gal drum. Run 6GC1 assumed breakup of the monolith into pieces so that the surface area increased by a factor of 10. The sequence of assumptions continued up to Run 6GC5, in which the surface area was increased by a factor of 100,000. For the IEB, the dissolution-limited release model discussed in Section 6.2.3.4, which considered the surface area of the waste form, was used.

The results are shown in Table 10-8. In all cases, the predicted maximum EDE is well below the limit of 25 mrem/yr in the performance objective. The IEB FWF results show, as expected, an initial increase in dose as the surface area was increased, reaching a peak dose of 1.2 mrem/yr when the surface area was increased by a factor of 10,000 relative to the 55-gal drum shape. Increasing the surface area further to a factor of 100,000 relative to the shape also yielded a dose of 1.2 mrem/yr, showing that an asymptote had been reached where further degradation of the FWF no longer increased the dose.

The time of maximum dose decreased slightly as the surface area was increased. However, the decrease was too small to appear in the two significant digits to which the times of peak dose are reported.

Results for the PCC FWF in Table 10-8 show that an asymptote of 1.1 mrem/yr is reached similarly when the diffusion path length of the FWF is decreased by a factor of only 10 relative to the original shape. The slight difference between the 1.2-mrem/yr asymptote for IEB and the 1.1-mrem/yr asymptote for PCC is the result of differences between a diffusion-limited release model for porous FWFs, such as PCC, and a dissolution-limited release model for nonporous FWFs, such as IEB.

Table 10-8. Effects of FWF degradation on performance at Site B.

Run number	FWF	Dimensional change ^a	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
9	PCC	Original diffusion path length	0.78	7.0E+06
9C1	PCC	Original diffusion path length/10	1.1	7.0E+06
9C2	PCC	Original diffusion path length/100	1.1	7.0E+06
6	IEB	Original surface area	0.015	7.0E+06
6GC1	IEB	Original surface area × 10	0.052	7.0E+06
6GC2	IEB	Original surface area × 100	0.29	7.0E+06
6GC3	IEB	Original surface area × 1,000	0.65	7.0E+06
6GC4	IEB	Original surface area × 10,000	1.2	7.0E+06
6GC5	IEB	Original surface area × 100,000	1.2	7.0E+06

a. The size and shape of a 55-gal drum were retained for the FWF for all runs, with varying degrees of fracturing.

Again, the decrease in the time of maximum dose was too small to appear in the table.

Once the FWF is degraded to the point that most of the inventory is released from the FWF during the first few time steps of the source-term model, further degradation of the FWF will not increase the dose. Under these conditions, the dose is controlled by the processes along the flowpath (decay, adsorption, dispersion, etc.) rather than by the processes controlling the release rate of radionuclides from the FWF.

10.2.5 Effects of Water Infiltration Rate on Results

Parametric evaluations were performed to assess the effects that varying the rate of infiltration of water through the disposal unit and vadose zone would have on predicted maximum EDEs and arrival times at Sites A, B, D, and E (see Table 10-9). Site C was not considered because of its insignificant groundwater pathway.

Twelve simulations were performed as part of this analysis. Runs 1, 1IR1, and 1IR2 simulated the performance of a PCC FWF at Site A using infiltration rates of 1.32 cm/yr (the original case), 5 cm/yr, and 10 cm/yr, respectively. Runs 9, 9IR1, and 9IR2 simulated the performance of a PCC FWF at Site B using infiltration rates of 1.2 cm/yr (the original case), 5 cm/yr, and 10 cm/yr, respectively. Runs 13, 13IR1, and 13IR2 simulated the performance of a PCC FWF at Site D using infiltration rates of 40 cm/yr (the original case), 80 cm/yr, and 120 cm/yr, respectively. Runs 14, 14IR1, and 14IR2 simulated the performance of a PCC FWF at Site E using infiltration rates of 0.5 cm/yr (the original case), 5 cm/yr, and 10 cm/yr, respectively. The rationale for using these infiltration rates is discussed in Section 6.2.3.2.

In Table 10-9, the doses at Sites A and B are primarily from the Am-241 decay chain and result in maximum EDEs below the 25-mrem/yr limit in the performance objective. As expected, the doses increase with increasing infiltration rates, albeit on a diminishing-returns basis.

At Sites D and E, the doses actually decrease in most cases with increasing infiltration rate. At Site D, the dominant contributor in Runs 13, 13IR1, and 13IR2 is the U-233 decay chain. Since U-233 has a long half-life (157,000 yr), the U-233 does not generate significant decay products during the short travel times at Site D. The U-233 has passed by the well by the time substantial ingrowth occurs. Thus, with an increasing infiltration rate that tends to shorten travel times, the dose decreases.

At Site E, the dominant contributor for Run 14 is the U-233 decay chain. For Runs 14IR1 and 14IR2, the dominant contributor is the Am-241 decay chain. With the increased infiltration rate in Runs 14IR1 and 14IR2, U-233 again passes by the well, so the U-233 decay chain dose is lowered and the Am-241 decay chain becomes the dominant contributor to dose. The Am-241 dose increases with the increasing infiltration dose. The time of maximum dose is inversely proportional to the infiltration rate, because the vadose zone travel time is inversely proportional to the infiltration rate. However, this relationship is true only for a given decay chain. The relationship between infiltration rate and predicted maximum EDEs is nonlinear because of the complex relationship between travel time and radioactive chain decay.

Table 10-9. Effects of infiltration rate on performance of PCC at Sites A, B, D, and E.

Run number	Site	Infiltration rate (cm/yr)	Vadose zone travel time (yr)	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)	Dominant decay chain contributing to dose
1	A	1.32	933	3.1	2.9E+06	Am-241
1IR1	A	5	246	6.1	7.6E+05	Am-241
1IR2	A	10	123	6.8	3.8E+05	Am-241
9	B	1.2	400	0.78	7.0E+06	Am-241
9IR1	B	5	96	4.4	1.7E+06	Am-241
9IR2	B	10	48	5.7	8.4E+05	Am-241
13	D	40	4.52	1,100	9.2E+03	U-233
13IR1	D	80	2.3	970	7.5E+02	U-233
13IR2	D	120	1.5	920	6.9E+03	U-233
14	E	0.5	450	3,400	4.7E+02	U-233
14IR1	E	5	45	2,900	2.7E+05	Am-241
14IR2	E	10	22	3,000	1.3E+05	Am-241

10.2.6 Effects of Assumptions Concerning Vadose Zone Basalt on Results, Separately and Combined with Infiltration Rate

Groundwater flow and radionuclide transport through the fractured vadose zone basalts at Site A were simulated by a simple equivalent-porous-medium approach.

It is difficult to determine a priori whether the equivalent-porous-medium approach used here is realistic or conservative on balance. To address that question, additional simulations were

performed to determine the importance of unsaturated flow and transport through fractured geologic units, by ignoring the presence of the fractured basalt layers at Site A.

For this purpose, a run designated 1NB (no basalt) was compared with Run 1, which simulated the performance of a PCC FWF at Site A. Sites B, D, and E were not considered for this analysis because there is no vadose zone basalt. Site C was not considered because the groundwater pathway was insignificant.

Run 1 used the simple equivalent-porous-medium approach, whereas Run 1NB completely ignored travel time through vadose zone basalts by assigning a zero thickness to these units. Run 1NB assumed a total vadose zone thickness of 9 m (see Table 3-1), the sediment thickness at Site A.

The revised assumption concerning the vadose zone basalt resulted in a travel time to the aquifer of 48 yr, compared with 933 yr in Run 1. The results are shown in Table 10-10. The dose doubles from 3.1 to 6.2 mrem/yr, and the time to peak dose decreases from 2.9 million yr to 0.72 million yr.

Additional runs were performed that considered the simultaneous effects of ignoring vadose zone basalts and increasing the infiltration rate. Run 1IR2NB was performed using a PCC FWF and the combination of a high infiltration rate of 10 cm/yr at Site A (Run 1IR2) and no credit for vadose zone basalt travel time (Run 1NB). The results, shown in Table 10-10, indicate no substantial synergistic effects when assumptions of a high infiltration rate and no basalt travel time were combined in Run 1IR2NB. When the infiltration rate is raised to 10 cm/yr, the presence or absence of fractured vadose zone basalts does not affect predicted doses substantially. Comparison with Run 1 shows that the conservative assumptions of a high infiltration rate and no basalt travel time increase the predicted doses by approximately a factor of 2 to 2.5, whether these assumptions are applied individually or are combined.

In Run 1IR2NB, the peak dose occurs from the U-233 decay chain at a time of 140,000 yr. By comparison, the peak dose in Runs 1, 1IR2, and 1NB is attributed to the Am-241 decay chain.

Table 10-10. Combined effects of high infiltration rate and no credit for travel time through vadose zone basalt on performance at Site A.

Run number	Infiltration rate (cm/yr)	Credit taken for vadose zone basalt?	Vadose zone travel time (yr)	Maximum dose (mrem/yr)	Time of maximum dose (yr)
1	1.32	Yes	933	3.1	2.9E+06
1IR2	10	Yes	123	6.8	3.8E+05
1NB	1.32	No	48	6.2	7.2E+05
1IR2NB	10	No	6.3	7.7	1.4E+05

10.2.7 Effects of Contaminant Sorption in Source Zone on Results, Separately and Combined with Infiltration Rate and Vadose Zone Basalt

As discussed in Sections 6.3 and 6.4, the radionuclides were assumed to be retarded by physical and chemical interactions with the geologic media in the vadose zone and the saturated zone. In the source zone (disposal facility), however, it was conservatively assumed that there was no retardation ($K_d = 0$).

A limited number of simulations were performed to assess the effects of sorption of radionuclides on soil or other backfill material in the source zone. Two cases were considered for the amount of sorption. In the first case, termed source-zone sorption, it was assumed that the geologic material in the source zone was not selected with sorptive capabilities in mind. In such backfill material, the values of the K_d s were conservatively assumed to be one-tenth of their values in the native sediments. In the second case, called tailored backfill, it was assumed that the geologic material was selected with sorptive capabilities in mind. In such backfill material, the values of the K_d s were conservatively assumed to equal those of the native sediments.

To assess the effects of sorption in the source zone, eight additional runs, designated 1SZS, 9SZS, 13SZS, and 14SZS (source-zone sorption) and 1TBF, 9TBF, 18TBF, and 6TBF (tailored backfill), were performed and compared with previous Runs 1, 9, 13, 14, 18, and 6, which did not consider this sorption in the source zone. Runs 1, 1SZS, and 1TBF considered PCC FWF disposed of at Site A, as did Runs 11R2NB, 11R2NBSZS, and 11R2NBTBF. Runs 9, 9SZS, and 9TBF considered PCC FWF disposed of at Site B. Runs 13 and 13SZS considered PCC FWF disposed of at Site D. Runs 14 and 14SZS considered PCC FWF disposed of at Site E. Runs 18, 18TBF, 6, and 6TBF considered IEB FWF disposed of at Sites A and B, respectively. Once again, Site C was not considered because of its low infiltration rate and insignificant groundwater pathway.

The sorption coefficient (K_d) values used for the source zone in the SZS runs were conservatively taken to be one-tenth the values used in Tables 3-2, 3-3, and 3-4 for the vadose zone sediments. The values used for the source zone in the TBF runs were considered the same as used in Table 3-2 for the vadose zone sediments.

The results of these 17 runs are shown in Table 10-11. The doses decrease for Sites A and B if adsorption within the disposal unit is considered, although the time of maximum dose is not substantially affected. The maximum doses are approximately inversely proportional to the K_d values used for the disposal unit backfill. The K_d s in Run 1TBF are 10 times greater than in Run 1SZS. This results in the dose being lower by about a factor of 10. This same effect can be seen with Runs 9SZS and 9TBF. The effect of source zone sorption is very pronounced for the runs with an IEB waste form. Run 18TBF has a dose that is about 3 orders of magnitude less than Run 18. A similar effect is seen when comparing Runs 6 and 6TBF.

Source-zone sorption has little effect on the dose at Sites D and E. The radionuclide K_d s at these two sites are generally low, so taking one-tenth of the far-field K_d provides for very little additional retardation in the source zone. At Site E, the K_d for uranium is zero, so there is no effect from source-zone sorption.

Table 10-11. Effects of adsorption of radionuclides by backfill within the disposal unit.

Run number	FWF	Site	Infiltration rate (cm/yr)	Credit taken for vadose zone basalt?	Credit taken for source zone sorption?	Vadose zone travel time (yr)	Maximum dose (mrem/yr)	Time of maximum dose (yr)
1	PCC	A	1.32	Yes	No	933	3.1	2.9E+06
1SZS	PCC	A	1.32	Yes	Yes	933	$2.3 \times 10E-01$	2.9E+06
1TBF	PCC	A	1.32	Yes	Yes	933	$2.3 \times 10E-02$	2.9E+06
11R2NB	PCC	A	10	No	No	6.3	7.7	1.4E+05
11R2NBSZS	PCC	A	10	No	Yes	6.3	2.9	1.4E+05
11R2NBTBF	PCC	A	10	No	Yes	6.3	0.25	8.4E+05
9	PCC	B	1.2	Yes	No	400	0.78	7.0E+06
9SZS	PCC	B	1.2	Yes	Yes	400	$5.5 \times 10E-02$	7.0E+06
9TBF	PCC	B	1.2	Yes	Yes	400	$4.8 \times 10E-03$	7.0E+06
13	PCC	D	40	N/A	No	4.52	1,100	9.2E+03
13SZS	PCC	D	40	N/A	Yes	4.52	1,100	9.2E+03
14	PCC	E	0.5	N/A	No	450	3,400	4.7E+02
14SZS	PCC	E	0.5	N/A	Yes	450	3,400	4.7E+02
18	IEB	A	1.32	Yes	No	933	$6.7 \times 10E-02$	2.9E+06
18TBF	IEB	A	1.32	Yes	Yes	933	$3.7 \times 10E-05$	2.9E+06
6	IEB	B	1.2	Yes	No	400	$1.5 \times 10E-02$	7.0E+06
6TBF	IEB	B	1.2	Yes	Yes	400	$6.6 \times 10E-06$	7.0E+06

Two final groundwater simulations (Runs 1IR2NBSZS and 1IR2NBTBF) were performed for Site A; these combined the two conservative assumptions of high infiltration rate and no basalt travel time but also assumed sorption of radionuclides within the source zone. The dose was reduced to 2.9 mrem/yr when routine source zone sorption was considered and to 0.25 mrem/yr when tailored backfill was assumed, compared with 7.7 mrem/yr without sorption in Run 1IR2NB.

It can be concluded by comparing Runs 1IR2NBSZS and 1IR2NBTBF with Run 1 that source zone sorption is a somewhat more important factor than infiltration rate or vadose zone basalt travel time or even the two combined.

The principal contributor to dose for Sites A and B is generally the Am-241 decay chain. For Runs 1IR2NB and 1IR2NBSZS, the principal contributor is the U-233 decay chain.

10.2.8 Effects of Far-Field Sorption Coefficients on Results

The values of sorption coefficients for given species in a given disposal environment are subject to considerable uncertainty. Therefore, parametric evaluations were performed to assess the effects of K_d s along the flow paths on predicted EDEs and arrival times. The initial values of the K_d s are listed in Sections 3.7, 3.8, and 3.9.

Four sets of simulations were performed. Runs 1, 1K1, and 1K2 simulated the performance of PCC at Site A using the initial values of K_d (Run 1), those K_d s reduced by a factor of 3 (Run 1K1), and the K_d s reduced by a factor of 10 (Run 1K2). The second set of simulations consisted of Runs 9, 9K1, and 9K2, which simulated the performance of PCC at Site B using the same sequence of K_d values just discussed. Site C was not considered because of its insignificant groundwater pathway. The third and fourth sets of simulations consist of Runs 13, 13K1, and 13K2, and Runs 14, 14K1, and 14K2, which simulated the performance of PCC at Sites D and E, respectively, using the same sequence of K_d values just discussed.

The results of these twelve runs are shown in Table 10-12. The results show that both the dose and the time of peak dose are strongly affected by the choice of K_d values.

At Sites A and B, reduction of the initial K_d s by factors of 3 and 10 causes reductions in peak dose arrival times by almost exactly the same factors. At Sites D and E, the dose for the initial K_d is dominated by U-233; however, the other runs are dominated by Am-241 or Pu-239. For Runs 13K1 and 13K2, the Pu-239 decay chain dominates the dose, whereas for Runs 14K1 and 14K2, the Am-241 decay chain dominates the dose. For Runs 13K1 and 13K2 and for Runs 14K1 and 14K2, the arrival times are reduced by the same factors as the K_d s.

The predicted doses increase with decreasing K_d s in direct response to the decreased travel time, which reduces the time allowed for decay. Run 1K2 exceeded the 25-mrem/yr performance objective limit by 3 mrem/yr, and Run 9K2 was only 1 mrem/yr below the limit.

At Sites D and E, all of the doses exceeded the performance objective.

Table 10-12. Effects of reduced K_d values on performance.

Run number	FWF	Site	K_d values ^a	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)	Dominant decay chain contributing to dose
1	PCC	A	Initial values	3.1	2.9E+06	Am-241
1K1	PCC	A	Initial values /3	12	9.5E+05	Am-241
1K2	PCC	A	Initial values /10	28	2.9E+05	Am-241
9	PCC	B	Initial values	0.78	7.0E+06	Am-241
9K1	PCC	B	Initial values /3	7.4	2.3E+06	Am-241
9K2	PCC	B	Initial values /10	24	7.0E+05	Am-241
13	PCC	D	Initial values	1,100	9.2E+03	U-233
13K1	PCC	D	Initial values /3	1,800	8.3E+03	Pu-239
13K2	PCC	D	Initial values /10	6,800	2.9E+03	Pu-239
14	PCC	E	Initial values	3,400	4.7E+02	U-233
14K1	PCC	E	Initial values /3	5,400	8.8E+05	Am-241
14K2	PCC	E	Initial values /10	14,000	2.7E+05	Am-241

a. The initial values are the sediment K_d values described in Sections 3.7, 3.8, and 3.9.

10.2.9 Effects of Distance to Receptor Well on Results

The effects of the distance to the receptor well on performance were evaluated in Runs 1 and 1LDW, assuming a PCC FWF at Site A. Run 1 assumed that the receptor well was located 100 m (328 ft) from the downgradient edge of the facility, while Run 1LDW assumed that the well was located 215 km (134 mi) downgradient. At this location, the groundwater may discharge to springs or other surface-water bodies.

The results of the evaluation are presented in Table 10-13. The predicted doses for the well located 215 km downgradient are approximately 2,500 times smaller than the doses for the well located at 100 m. This is a direct result of the considerable dispersion that occurs along the migration pathway and the radionuclide decay that occurs during the longer travel times.

10.2.10 Effects of Untreated Waste on Results

The other portions of Section 10 discuss the performance of various treated FWFs. This section presents the results of additional evaluations that predict the performance of untreated waste. The results are compared against those for a PCC FWF. Although the ALLW in storage at the INEL will probably be immobilized by some method before disposal, the projected doses from untreated waste serve as a point of a comparison against which the relative effectiveness of proposed waste treatment options can be judged.

For the untreated waste, the total refined inventory was divided into three categories, as discussed in Section 2.4.5 and Appendix B: sludge, nonsludge, and RH waste. The specific radionuclide inventories and volumes associated with each of the three categories are given in Table 2-11.

The radionuclide inventory associated with the sludge fraction was assumed initially to be released by diffusion, and the inventories associated with the nonsludge and RH fractions were assumed to be released by a surface washoff mechanism. Additional runs were performed that (a) varied the diffusion constants for releases from the sludge, (b) considered release of the entire inventory by a diffusion mechanism, and (c) considered release of the entire inventory by surface washoff.

The sludge fraction, which occupies 16% of the total waste volume, consists of a wide variety of cemented and noncemented chemical precipitates (oxides, hydroxides, oxyhydroxides, and nitrate salts) in a solid or semisolid form. The radionuclides are present in these sludges as either soluble salts or as ions dissolved in the residual pore water.

Table 10-13. Effects of distance to receptor well on performance.

Run number	FWF	Distance to receptor well	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
1	PCC	100 m	3.1	2.9E+06
1LDW	PCC	215 km	1.2 x 10E-03	3.4E+06

The nonsludge and RH fractions, which together occupy 84% of the total waste volume, consist mostly of common materials such as paper, plastic, wood, sheet metal, plexiglass, tools, and laboratory glassware that have surface contamination. Radionuclides are released from the surface of these materials by surface washoff in water that infiltrates through the disposal unit.

The results for untreated waste at Sites A (Run 1UTR), B (Run 9UTR), D (Run 13UTR), and E (Run 14UTR) are shown in Table 10-14 along with the results of Run 1 (PCC at Site A), Run 9 (PCC at Site B), Run 13 (PCC at Site D), and Run 14 (PCC at Site E) for comparison. The predicted peak annual dose from the untreated waste at Site A was 3.6 mrem/yr, which is slightly higher than the predicted dose (3.1 mrem/yr) from the PCC FWF at Site A. At Site B, the predicted maximum dose from the untreated waste was 0.92 mrem/yr, which is slightly higher than the dose from PCC FWF at Site B. The increases for Sites A and B were a result of the higher radionuclide release rates assumed for the nonsludge and RH fractions.

The dose at Site D from untreated waste is 780 mrem/yr; this is less than the dose from the PCC FWF at this site. The dose from untreated waste is lower because the volume of untreated waste is greater than the volume of PCC waste. Since the volume is greater, the facility is larger and its center is farther from the well. This results in longer travel times and more radioactive decay.

Table 10-14. Effects of disposing of untreated waste.

Run number	FWF	Site	Maximum dose (mrem/yr)	Time of maximum dose (yr)
1	PCC	A	3.1	2.9E+06
1UTR	Untreated	A	3.6	2.9E+06
1UTR TORT	Untreated	A	3.7	2.9E+06
11R2NB	PCC	A	7.7	1.4E+05
11R2NBUTR	Untreated	A	8.5	1.4E+05
9	PCC	B	0.78	7.0E+06
9UTR	Untreated	B	0.92	7.0E+06
13	PCC	D	1,100	9.2E+03
13UTR	Untreated	D	780	9.5E+03
14	PCC	E	3,400	4.7E+02
14UTR	Untreated	E	7,000	4.7E+02

At Site E, the dose from untreated waste is 7,000 mrem/yr, more than double the dose from PCC waste. The dose from untreated waste is high because of the short travel times at Site E. The initial release of waste resulting from surface washoff does not get diluted over this relatively short travel time. Also, the K_d s at Site E are low, so the facility size has little effect on dose.

A similar comparison was made for the situation in which more conservative assumptions were used for the infiltration rate and the basalt travel time. Run 1IR2NB used a PCC FWF, with a high infiltration rate of 10 cm/yr and no credit for vadose zone basalt travel time. The groundwater travel time with the higher infiltration rate and without credit for the basalt was 6.3 yr, compared with 933 yr with basalt and the lower infiltration rate. Run 1IR2NBUTR was the same as Run 1IR2NB but the waste was untreated. Again, the untreated run gives a slightly higher dose than the corresponding PCC case: 8.5 mrem/yr versus 7.7 mrem/yr.

Run 1UTR assumed that the sludge had diffusion coefficients equal to those of a PCC waste form. Run 1UTRTORT used a comparative tortuosity of 1.0, which makes the diffusion coefficient the same value as that for the radionuclide diffusing in free water. No credit is taken in this case for any retention by the sludge. The groundwater scenario is not very sensitive to diffusion coefficients in this range, and the maximum dose approaches an asymptote as the diffusion coefficient increases. The higher diffusion coefficient case (Run 1UTRTORT) gives only a slightly higher dose than Run 1UTR: 3.7 versus 3.6 mrem/yr. The predicted doses for untreated waste and PCC are both an order of magnitude less than the performance objective of 25 mrem/yr.

Since the characteristics of the untreated waste are uncertain, additional runs were performed to examine the sensitivity of the dose to two different release mechanisms: diffusion and surface washoff. Runs 1UTRDIFF and 1UTRSUR, respectively, simulate these two mechanisms. For Run 1UTRDIFF, the radionuclides in all of the waste were assumed to be released by diffusion. The diffusion coefficient used was the same as that for PCC. For Run 1UTRSUR, the radionuclides in all of the waste were assumed to be released by surface washoff. The results are shown in Table 10-15. The maximum EDEs are not very sensitive to the different mechanisms. Surface washoff gives the highest dose because most of the inventory is available for immediate release. Diffusion gives a lower dose because the release is spread out over a longer time period than in the surface washoff run. As shown in Table 10-15, the doses fall on either side of those for Run 1UTR, which uses a combination of these two release mechanisms.

Table 10-15. Effects of assumed release mechanism for untreated waste.

Run number	FWF	Mechanism	Maximum dose (mrem/yr)	Time of maximum dose (yr)
1UTRDIFF	Untreated	Diffusion	2.8	2.9E+06
1UTRSUR	Untreated	Surface washoff	3.7	2.9E+06
1UTR	Untreated	Surface washoff plus diffusion	3.6	2.9E+06

10.2.11 Effects of FWF Leach Rates on Results

An extensive amount of laboratory leach rate data exists for a variety of glass and ceramic FWF compositions. A common characteristic of these laboratory data is a large amount of scatter in test results. The scatter is a result of several factors including:

- Variations in FWF composition (alkali content, waste loading, etc.)
- Differences in FWF processing conditions (melt temperature, cool-down rate, annealing temperature, etc.)
- Differences between leach test protocols (temperature; surface area; leachant composition; static, dynamic, or flow-through test conditions; etc.).

Selecting the most appropriate data on which to base assigned leach rates in PA runs is a challenging task. The leach rates used for most of the PA runs were obtained from a survey of existing 28-day MCC-1 leach rate data, predominantly from Lutze and Ewing (1988) with the exception of bulk leach rates for IEB4, the more favorable ceramic. The leach rates appear in Table G-1.

Recent experiments at the INEL have provided new leach rate data for three FWFs. See the January 18, 1995, letter from T. L. Eddy in Appendix G. The new data are tabulated in Table 10-16. The new values, referred to as "revised leach rates," are slightly larger than the previous values. The revised values are based on the bulk release rate of silica during MCC-1 leach testing of ASG, IEB, and IEB4 FWFs. The leach rates for the radionuclides are typically smaller than the bulk or silica leach rate by a substantial margin, so use of the indicated values is conservative. The availability factors for ASG and IEB remain at 100% and the availability factor for IEB4 remains at 30%.

Six sets of evaluations were performed to simulate the long-term performance of three FWFs with the revised leach rates. These sets were for ASG, IEB, and IEB4 at Sites A and B.

The results for the runs with the revised leach rates (designated by run numbers ending in RLR) are as shown in Table 10-17. In all cases, the predicted peak doses were dominated largely by the Am-241 decay chain. The Pu-238 and Pu-239 decay chains were also contributors, but yielded much smaller doses than the Am-241 chain and the peaks occurred at a much later time. The results were compared with the doses obtained for the original leach rates. The peak doses showed a slight increase because of the small increase in the leach rates for the three FWFs. This result is expected because the radionuclide release rate increases with an increase in leach rate. The time for maximum dose did not change, because the leach rate would affect only the concentration of radionuclides released and would not change the time of peak occurrence at the receptor well.

All of the indicated doses are at least 2 orders of magnitude below the performance objective of 25 mrem/yr. The doses calculated with the revised leach rates differ by no more than a factor of 2 from the corresponding doses calculated with the original leach rates. Therefore, the precise value of the leach rate for these three FWFs does not appear to be an important

Table 10-16. Leach rates for the final waste forms.

Final waste form	Original leach rate ^a (g/m ² -day)	Revised leach rate ^b (g/m ² -day)
ASG	0.07	0.13
IEB	0.08	0.10
IEB4	0.05	0.11

a. From Table G-1.

b. From the January 18, 1995, letter by T. L. Eddy in Appendix G.

Table 10-17. Effects of leach rates on performance.

Run no.	Final waste form	Site	Leach rate ^a (g/m ² -day)	Peak dose (mrem/yr)	Time of peak dose (10 ⁶ yr)
17	ASG	A	0.07 (b)	0.066	2.9
17RLR	ASG	A	0.13 (r)	0.12	2.9
18	IEB	A	0.08 (b)	0.067	2.9
18RLR	IEB	A	0.10 (r)	0.083	2.9
3	IEB4	A	0.05 (b)	0.011	2.9
3RLR	IEB4	A	0.11 (r)	0.023	2.9
12	ASG	B	0.07 (b)	0.015	7.0
12RLR	ASG	B	0.13 (r)	0.027	7.0
6	IEB	B	0.08 (b)	0.015	7.0
6RLR	IEB	B	0.10 (r)	0.02	7.0
8	IEB4	B	0.05 (b)	0.0024	7.0
8RLR	IEB4	B	0.11 (r)	0.0054	7.0

a. b = original value; r = revised value.

consideration in selecting a FWF, under the assumptions used herein and for Sites A and B. The same general conclusion holds for the BSG FWF, for which no new leach rate data were identified.

10.2.12 Effects of Time-Dependent Leach Rates on Results

Leach rates for FWFs are time-dependent. When glass or ceramic FWFs are initially in contact with water, the leach rate is at a maximum value. As leaching proceeds, the concentrations of the major constituents in the waste form increase in the leachant, which reduces the driving force for dissolution. In the case of BSG, for example, as the concentration of silica in the leachant approaches the solubility of amorphous silica, the dissolution rate of the waste form approaches a very low value. In some long-term tests, secondary phases such as cristobalite (SiO_2) and zeolites (hydrated Na-Ca-aluminosilicates) have been observed to precipitate from the leachant. These precipitates act to buffer the silica concentration at a constant value corresponding to the solubility limit of the secondary phase assemblage. Under these conditions, the release rate of radionuclides from the glass is indirectly controlled by the rate of growth of the secondary phases and is also affected by the influx rate of fresh water, which dilutes the saturated leachant.

Leaching of glass can thus be considered as a two-stage process. Stage one proceeds until the silica concentration approaches a saturated value and is characterized by a relatively high release rate. Stage two proceeds from this point until the waste form is completely dissolved and is characterized by a lower steady-state rate. Ceramics generally exhibit similar behavior.

The use of short-term leach rate to model releases occurring over many thousands of years can be quite conservative. Therefore, leach rate data were examined so that a two-stage release model for glasses and ceramics could be used in a limited number of runs.

A March 14, 1995, letter from T. L. Eddy (see Appendix G) documents the basis for recommending the values for the two-stage leach rate model. The recommended values are given in Table 10-18.

PA runs with the two-stage leach model were made only for the IEB FWF. For conservatism, the Stage 2 leach rate value selected for the IEB runs matched that for the FWF with the largest value for Stage 2 leaching, namely, the value of $0.0025 \text{ g/m}^2\text{-day}$ for BSG.

Table 10-18. Recommended and selected values for the two-stage leach rate model, $\text{g/m}^2\text{-day}$.

Final waste form	Stage 1 $0 < t < 1 \text{ yr}$	Stage 2 recommended $t > 1 \text{ yr}$	Stage 2 selected $t > 1 \text{ yr}$
BSG	0.4	0.0025	No runs
ASG	0.07	0.00044	No runs
IEB	0.08	0.00050	0.0025
IEB4	0.05	0.00031	No runs

Two simulations were performed to evaluate the effects of a two-stage leach process. The performance of IEB at Sites A and B was evaluated using a 0 to 1-yr leach rate corresponding to the baseline value used in previous runs, followed by the lower leach rate as shown in Table 10-18. Because of the large uncertainties in long-term leach rates, the results should be viewed as illustrative rather than definitive.

Results of the time-dependent leach rate simulations (Runs 18TDLR and 6TDLR) are shown in Table 10-19, along with the corresponding fixed-leach-rate runs (numbered 18 and 6) for comparison. The doses at both Sites A and B are reduced by more than an order of magnitude.

Doses at both sites are dominated by the lower leach rate because of the relatively short duration of the high-leach-rate period. The pulse of radionuclides released during this 1-yr period is dispersed along the travel path, so the peak is attenuated before arrival at the receptor well. There probably exists some critical duration of the period with the initial high leach rate, beyond which the doses are dominated by the higher rather than the lower leach rate. This critical duration is apparently >1 yr at Sites A and B.

10.2.13 Effects of Chemical Interactions within Cement Final Waste Forms on Results

Nearly all of the runs that evaluated the performance of cement FWFs considered only the physical retardation of radionuclides within the FWF and did not consider the effects of chemical retardation (see Section 6.2.3.3). This section considers chemical as well as physical retardation within the cement matrix and the consequent effects on the predicted doses.

The release of radionuclides from a cement waste form will be largely diffusion-controlled as long as the cement is intact. For the purposes of these analyses, it was assumed that the cement is intact until all of the radionuclides are released by diffusion. For the runs without chemical interaction, the radionuclide inventory is released from the cement matrix in about 5,500 yr. For the runs with chemical interaction, the nuclides are released by a combination of diffusion and desorption in about 100,000 yr. The estimated life of the cement is uncertain. It depends on the waste, treatment parameters, residual stresses, temperature and moisture cycles, and groundwater chemistry. Published estimates (e.g., Brandstetter et al. 1994) suggest the initiation of substantial cracking in 100 to 1,000 yr, and large-scale breakup in 500 to 5,000 yr. Even with large-scale breakup, however, the concrete pieces could provide considerable retardation of radionuclides.

Table 10-19. Effects of time-dependent FWF leach rates on performance.

Run no.	Final waste form	Site	Leach rate (g/m ² -day)	Peak dose (mrem/yr)	Time of peak dose (10 ⁶ yr)
18	IEB	A	0.08	0.067	2.9
18TDLR	IEB	A	Time-dependent	0.0021	2.9
6	IEB	B	0.08	0.015	7.0
6TDLR	IEB	B	Time-dependent	4.7 × 10E-04	7.0

Nevertheless, the 100,000 yr required for the last radionuclides to exit the cement FWF, when credit is taken for chemical interactions, is far beyond the expected life of the cement. Thus, the accompanying analysis of chemical interactions may provide a lower bound on the dose, in that credit is taken for the concrete structure long after it may have eroded or crumbled into small pieces.

The radionuclide migration behavior within the cement matrix will be strongly affected by the chemical composition of the pore water within the matrix. Critical pore water parameters include pH, redox, and ionic strength. The chemical environment within portland-type cement/concrete is characterized by high pH and high total dissolved solids. The pH can range from 11.5 for a low-alkali formulation to over 13 for high alkali formulations. This high pH is maintained by the presence of portlandite [$\text{Ca}(\text{OH})_2$], which will buffer the pH above 11.5 for as long as the portlandite remains within the matrix. The presence of Na_2O and K_2O , which hydrolyze to NaOH and KOH in the matrix environment, will raise the pH to values above the portlandite buffer (Glasser et al. 1985).

Published information on chemical retardation in cement waste forms, such as Allard (1985), provides values that are calculated from laboratory data. These calculated values lump all sorption and precipitation processes together and express the net effects of these processes as a K_d without distinguishing between the exact mechanisms. This approach does not explain the mechanisms responsible for chemical retardation, but it is adequate for assessing the general effects of chemical retardation.

The solubilities of the actinide elements are several orders of magnitude lower at these high pH conditions than at neutral pH values. Radionuclides released from the FWF matrix will encounter high pH pore fluid and precipitate as hydroxide or oxyhydroxide phases (Allard 1985). The presence of these phases will maintain the concentrations of actinides in the pore fluid at low values corresponding to the solubility limits of the precipitated phases. This control on the release rate of actinides will persist until either the radionuclides or the pH-controlling phase is completely leached. Adsorption of radionuclides within the cement matrix can also occur, which can reduce the effective rate of diffusion through the waste form. Specific sorption processes such as chemisorption, electrostatic interaction, and ion exchange can contribute to chemical retardation. Precipitation or solubility control is not, strictly speaking, a sorptive process, but its effects can be treated, from a modeling standpoint, as retardation.

Based on the aforementioned assumptions, Equation 6-5 was used to determine the effective diffusion coefficients of the radionuclides within the cement waste forms:

$$D_i^e = \frac{D_i^o}{R_f \tau} \quad (6-5)$$

where

D_i^e = effective diffusion coefficient of radionuclide i (cm^2/sec)

D_i^o = diffusion coefficient of radionuclide i in water, (cm^2/sec)

R_f = chemical retardation factor (dimensionless)

τ = physical retardation factor or tortuosity (dimensionless).

The physical retardation factor was assumed to be 200 for the PCC FWF (see Section 6.2.3.3). The chemical retardation factor was calculated using K_d s for radionuclides in a cement matrix (Allard 1985). Equation 10-1 was used to determine the retardation coefficient R_f :

$$R_f = 1 + (\rho/\phi) K_d \quad (10-1)$$

where

ρ = density of the waste form = 1.95 g/cm^3 for PCC

ϕ = porosity of the waste form = 0.22 for PCC.

The K_d value used for americium, plutonium, and uranium is $5,000 \text{ cm}^3/\text{g}$ (Allard 1985), yielding a chemical retardation of 44,300. The effective diffusion coefficients are thus reduced by this same factor.

For PCC at Sites A and B, Table 10-20 shows the results of runs with chemical interactions, along with the runs that included only physical retardation due to the tortuosity of the diffusion paths in the cement. The results indicate that the doses decrease by approximately 2 orders of magnitude when chemical interactions within the cement waste form are considered. In fact, the doses in runs 1INT and 9INT are within a factor of three of the corresponding doses for the IEB4 FWF in Runs 3 and 8, respectively.

Table 10-20. Effects of chemical interactions of radionuclides with cement.

Run number	FWF	Site	Physical interaction considered?	Chemical interaction considered?	Maximum EDE (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
1	PCC	A	Yes	No	3.1	2.9E+06
1INT	PCC	A	Yes	Yes	0.025	2.9E+06
9	PCC	B	Yes	No	0.78	7.0E+06
9INT	PCC	B	Yes	Yes	0.0066	7.0E+06

10.2.14 Effects of Refined Inventory on Results

During the second and third phases of the investigation, evaluations were performed in which the total waste volume and radionuclide inventory were subdivided into a CH fraction and an RH fraction. This refined inventory is discussed in Section 2.4.3 and in Section A-9 of Appendix A.

Evaluations were performed assuming an IEB FWF disposed of at Site B. These evaluations were identical to Run 6, except that separate evaluations were performed for the CH and RH fractions of the waste. Additional evaluations were performed assuming PCC and IEB FWFs disposed of at Site D.

Detailed evaluations of groundwater transport for these cases at Site B were not necessary, however, for the following reason. The dominant radionuclides at Site B were determined in previous runs to be the actinides. In deriving the refined inventory in Appendix A, the volumetric concentration of actinides was assumed to be identical in both the CH and RH fractions, and the RH fraction occupies only 31 m³, or 0.13% of the total combined inventory volume. Therefore, predicted doses from the groundwater scenario are highly insensitive to the RH waste, which contains most of the nonactinide activity. The exclusion or segregation of the RH waste fraction from the CH waste fraction does not affect the predicted doses from groundwater transport at Site B because the long travel times provide ample time for the RH nonactinides to decay before reaching the receptor well.

Detailed evaluations of groundwater transport were performed for the CH and RH fractions disposed of as PCC and IEB at Site D because the shorter groundwater travel time at SRS raised the possibility that doses from the nonactinides in the RH fraction might be significant. Although groundwater travel times at SRS are shorter than the groundwater travel times at the semiarid sites, the use of higher site-specific adsorption coefficients for several nonactinides (e.g., Sr-90 and Cs-137) at SRS relative to those at the semiarid sites resulted in long (~20,000-yr) radionuclide travel times. Under these conditions, the RH nonactinides have ample time to decay before reaching the receptor well. As a result, the doses at Site D from the RH fraction are approximately 0.1% of the doses from the CH fraction, which corresponds to the RH-versus-CH volumetric percent.

It can be concluded from these evaluations that the inclusion or deletion of the RH fraction of the ALLW waste has an inconsequential effect on the doses from the groundwater scenario, as long as the radionuclide travel times at the site are long enough to allow for substantial decay of the shorter-lived RH nonactinides. Thus, no nonintrusion results showing the effects of the refined inventory are included in Table 10-2.

10.2.15 Effects of Inventory with Additional Refinements on Results

In the inventory with additional refinements, four nonactinides (H-3, C-14, Tc-99, and I-129) were not screened out from detailed groundwater transport calculations (see Sections 2.4.4 and 2.5.4). In groundwater transport calculations involving the preceding two inventories, the nonactinides were screened out.

Additional groundwater simulations were performed that included these nonactinides in addition to the actinide inventory. Runs 1, 13, 14, 4A, and 1UTR do not include the nonactinide inventory, whereas Runs 1IAR, 13IAR, 14IAR, 4AIAR, and 1UTRIAR include the four nonactinides listed above.

The results of these runs are shown in Table 10-21. Between Runs 1 and 1IAR, the inventory with additional refinements produces about a 1.0% increase in dose; between Runs 13 and 13IAR, a 0.7% increase; between Runs 14 and 14IAR, a 0.5% increase. These increases are not evident in Table 10-21 because the results are reported to only two significant figures. Between Runs 1UTR and 1UTRIAR, the nonactinides provide a 3.0% increase in dose.

In all of these runs, the actinides dominate the dose. Of the nonactinides, C-14 makes the greatest contribution to dose. In Runs 1IAR, 14IAR, and 1UTRIAR, C-14 has a sorption coefficient of 0.0, which results in a retardation coefficient of 1.0. This lack of retardation results in relatively short travel times for C-14. Because it has low concentrations (1.18×10^{-9} Ci/m³ in PCC) in the FWFs, I-129 contributes very little dose. Tc-99 also has a relatively low concentration (3.27×10^{-6} Ci/m³ in PCC) in the FWFs. H-3 has retardation coefficients that are about the same as those for C-14, but H-3 has a short (12.3 yr) half-life, whereas the half-life of C-14 is 5,730 yr. For Run 13IAR, C-14 has a retardation coefficient of 43, but C-14 still dominates the nonactinides because it has a relatively high (when compared to the other nonactinides) concentration (6.54×10^{-5} Ci/m³ in PCC) in the FWFs. This higher retardation coefficient for C-14 is based on sorption coefficients from Martin Marietta (1994).

Table 10-21. Effects of inventory with additional refinements on performance.

Run number	FWF	Site	Maximum dose (mrem/yr) limit = 25 mrem/yr	Time of maximum dose (yr)
1	PCC	A	3.1	2.9E+06
1IAR	PCC	A	3.1	2.9E+06
13	PCC	D	1,100	9,200
13IAR	PCC	D	1,100	9,200
14	PCC	E	3,400	470
14IAR	PCC	E	3,400	470
4A	IEB4	C	3.5×10^{-4}	4.3E+07
4AIAR	IEB4	C	0.75	3,200
1UTR	Untreated	A	3.6	2.9E+06
1UTRIAR	Untreated	A	3.7	2.9E+06

Between Runs 4A and 4AIAR, the increased dose is dominated by the nonactinides because of their short travel times. Run 4AIAR is the only run in which the nonactinides dominate the total dose. At Site C, the shortest actinide travel time is about 35 million yr, compared with the shortest nonactinide travel time of about 3,200 yr. Because of the reasons given in the previous paragraph, C-14 dominates the dose for Run 4AIAR.

Thus, the nonactinides, especially C-14, contribute to total dose, but the contribution to dose is overshadowed by the contribution from the actinide inventory. The one exception is at Site C, where the actinides have extremely long travel times compared with the nonactinide travel times.

The predicted contributions from C-14, Tc-99, and I-129 should be considered upper bounds because it is unlikely that these nuclides will remain in any of the FWFs (other than untreated waste) considered in this report. The cement, glass, and ceramic waste forms result from the cementation, vitrification, or crystallization of ash from the thermal treatment of the waste. The volatile nature of these elements makes it likely that they will partition into the off-gas system.

10.2.16 Effects of Revised Uranium and Thorium Inventories on Results

The simplified actinide inventory used in virtually all runs in this study (Table 2-3) contained 13 Ci of U-233 and no Th-232. Recent data from Apel et al. (1994) suggest the presence of 865 Ci of U-233 and 2.5 Ci of Th-232 (see Section 2.4.2.5).

The larger values for U-233 and Th-232 were used in Runs 1UTH and 3UTH to assess the potential importance of these two radionuclides in FWFs. These additional runs calculated doses from the disposal at Site A of a PCC FWF (Run 1UTH) and an IEB4 FWF (Run 3UTH), both of which contained the higher U-233 and Th-232 inventories. The results were then compared with the performance of these two FWFs using the original actinide inventory.

The total activity of Am-241 used in both inventories for this particular simplified comparison was 3,741 Ci. [The 3741 Ci includes the 1340 Ci of Am-241 and the 2401 Ci of its comparatively short-lived parent Pu-241. Because the half-life of Pu-241 is much less than that of Am-241, it can be shown (Glasstone and Sesonke 1967, for example) that the addition of the two activities is a valid approximation in this case.] Each radionuclide was run separately in order to compute the dose attributable to each parent radionuclide initially present in the inventory.

The predicted doses for the groundwater pathway based on the higher actinide inventory are shown in Table 10-22. The doses for Run 1 (PCC at Site A) and Run 3 (IEB4 at Site A), both of which used the simplified actinide inventory, are listed for comparison.

For Run 1UTH, the maximum doses for U-233 and Th-232 were approximately 5 orders of magnitude lower than the dose for Am-241. For Run 3UTH, the maximum dose for U-233 was again about 5 orders of magnitude less than the Am-241 dose, and the maximum dose for Th-232 was about 3 orders of magnitude less than the Am-241 dose.

Table 10-22. Effects of possible increases in U-233 and Th-232 inventory on performance.

Run number	FWF	Site	Maximum dose (mrem/yr)	Time of maximum dose (yr)	Decay chains contributing to dose	Total inventory of each radionuclide (Ci)	Maximum dose from each decay chain (mrem/yr)	Total dose (%)
1	PCC	A	3.1	2.9E+06 4.1E+06 NA	Am-241 U-233 Th-232	3741 ^a 13 0	3.1 — ^b 0	100
1UTH	PCC	A	3.1	2.9E+06 4.1E+06 4.1E+06	Am-241 U-233 Th-232	3741 ^a 865 2.5	3.1 1.4E-05 9.1E-05	100
3	IEB4	A	0.011	2.9E+06 4.1E+06 NA	Am-241 U-233 Th-232	3741 ^a 13 0	0.011 — ^c 0	100
3UTH	IEB4	A	0.011	2.9E+06 4.1E+06 4.1E+06	Am-241 U-233 Th-232	3741 ^a 865 2.5	0.011 2.6E-07 1.7E-05	100

a. See text for explanation of Am-241 inventory.

b. Dose at 2.9E+06 yr, when the highest total peak occurs, is essentially 0. Dose at 4.1E+06 yr, when the U-233 peak occurs, is 2.1E-07 mrem/yr.

c. Dose at 2.9E+06 yr, when the highest total peak occurs, is essentially 0. Dose at 4.1E+06 yr, when the U-233 peak occurs, is 3.9E-09 mrem/yr.

In both Run 1UTH and Run 3UTH, the Am-241 contributed 100% of the total dose when rounded to the nearest percent. In both runs, the maximum doses for U-233 and Th-232 occurred at 4.1 million yr, compared with 2.9 million yr for Am-241. This is because U-233 and Th-232 have the same K_d value and will arrive at the receptor well at the same time, whereas the K_d value for Am-241 is lower, resulting in an earlier arrival time.

In summary, the predicted doses were unchanged by the assumed potential increases of U-233 and Th-232.

10.2.17 Effects of Vault-Type Disposal Facility on Results

Analyses were performed to assess the long-term performance of a vault-type disposal facility at the SDA (Site A), Site 14 (Site B), and Hanford (Site E). Preliminary design plans are based on the Center de Stockage de l'Aube disposal facility in France, which is a concrete vault facility with waste packages stabilized with concrete or gravel backfill, and a multi-layered barrier for a closure cap.

The important parameters considered for the analyses are the facility lifetime and the infiltration rate during the facility lifetime. The design lifetime for the l'Aube facility is 300 yr after closure, during which time the facility remains intact. The requirements established by the NRC in the regulations of 10 CFR 61 include waste form stability for 300 yr and intrusion protection for 500 yr. Based on these requirements, a facility lifetime of 500 yr was assumed, during which time the facility remains intact. While the facility is intact, the infiltration rate was assumed conservatively to be reduced by roughly a factor of 10. After the facility lifetime of 500 yr, it was assumed that the facility instantly fails, and the infiltration rate returns to the normal value. For Sites A and B, the infiltration rate with the intact facility was 0.1 cm/yr. The infiltration rates at Sites A and B after the failure of the facility were 1.32 and 1.2 cm/yr, respectively. At Site E, the infiltration rate prior to facility failure was 0.05 cm/yr; after failure, it was 0.5 cm/yr. Simulations were performed for PCC and IEB FWFs.

The assumptions used here for vault lifetime and infiltration rates differ somewhat from those in Section 5.2 that were used to evaluate the conceptual disposal of the INEL ALLW at Site D. For Site D, the present study simply followed the assumptions in the Site D PA (Martin Marietta 1994). In this section, the vault lifetime assumption is based on compliance with the NRC guidelines. The infiltration rate assumption was a conservative estimate based on professional judgment.

The results for runs 1VTF, 9VTF, 14VTF, 18VTF, 6VTF, and 19VTF with the vault type facility are provided in Table 10-23 and compared with the corresponding Runs 1, 9, 14, 18, 6, and 19. Runs 1, 1VTF, 9, 9VTF, 14, and 14VTF considered PCC FWFs for Sites A, B, and E, respectively. Runs 18, 18VTF, 6, 6VTF, 19, and 19VTF considered IEB FWFs for Sites A, B, and E respectively.

For the PCC FWF, the reduction in dose offered by the vault barrier is minimal (a few percent) at sites that have relatively long travel times for radionuclides. The only site where a significant reduction in dose (38%) was predicted was Site E, where the radionuclide travel times are relatively short.

Doses for IEB at the sites evaluated were essentially unchanged by the presence of a vault barrier. For the IEB FWF, the releases occur over longer time periods than for PCC, so the doses are unaffected by changes in the release rate during the first 500 yr after closure. Runs 18VTF and 6VTF have slightly higher doses than Runs 18 and 6, respectively. The small increase in dose is due to a very small increase in travel times, caused by the presence of the vaults. The increase in travel time allows more decay of the Am-241 chain into decay products with higher dose-conversion factors.

Separate intruder analyses were not performed with a vault-type facility because of the following rationale. For any scenario initiated by drilling, it was assumed that the drill bit would bore through the vault and exhume the waste. This assumption was also applied to any scenario initiated by excavation; hence, the intruder would excavate the waste regardless of the presence of the vault. Also, the intruder analyses assumed that there was no leaching of radionuclides out of the facility, so the vault would not affect the release of radionuclides during the intruder scenarios.

10.2.18 Reasons for Large Doses at Sites D and E

10.2.18.1 Introduction. All of the groundwater transport runs involving hypothetical disposal of the INEL ALLW at Site D or E predict a much larger dose than for disposal of the same waste at Site A, B, or C. For corresponding cases, the doses at Sites D and E are typically 2 to 4 orders of magnitude larger than at Sites A and B and sometimes even larger. Comparing the doses at Sites D and E against the corresponding doses at Site C results in still larger differences. In addition, the doses at Sites D and E are above the limit in the performance objective in all cases but one: IEB4 at Site E.

The question arises as to why these doses for the INEL ALLW at Sites D and E are so high, in spite of the fact that

- Where feasible, the parameter values used herein for Sites D and E were those used in the respective PAs prepared for those sites for the disposal of their own LLW
- Those PAs concluded that the disposal of the waste generated at their own sites would comply with the performance objectives.

The remainder of Section 10.2.18 is directed toward answering that question.

This section explores some of the reasons for the much higher calculated doses at Sites D and E. Discussions with a preparer of the Hanford PA indicate that still other factors are affecting the calculated doses, e.g., inhomogeneities of contaminant distribution within the waste coupled with solubility limits for dose-controlling radionuclides. Only a point-by-point comparison of the two methods would reveal all of the important differences, which is beyond the scope of this study.

10.2.18.2 Comparisons of Results from Runs Already Made. To explore the above question, several key runs were examined that had already been made for waste disposal at Sites D and E. Some of the runs were for INEL ALLW, as evaluated in this study. The other runs

Table 10-23. Effects of vault-type facility for given disposal sites.

Run number	FWF	Site	Maximum dose (mrem/yr)	Time of maximum dose (yr)	Dominant contributor to dose (decay chain)
1	PCC	A	3.1	2.9E+6	Am-241
1VTF	PCC	A	2.9	2.9E+6	Am-241
9	PCC	B	0.78	7.0E+6	Am-241
9VTF	PCC	B	0.72	7.0E+6	Am-241
14	PCC	E	3,400	470	U-233
14VTF	PCC	E	2,100	470	U-233
18	IEB	A	0.067	2.9E+6	Am-241
18VTF	IEB	A	0.068	2.9E+6	Am-241
6	IEB	B	0.015	7.0E+6	Am-241
6VTF	IEB	B	0.017	7.0E+6	Am-241
19	IEB	E	150	22,000	U-233
19VTF	IEB	E	150	22,000	U-233

were for specific inventories of SRS LLW and Hanford LLW disposed of at their own sites, as evaluated in their respective PAs (Martin Marietta 1994 and Wood et al. 1994).

The results of these comparisons appear in Table 10-24. The objective of the table was to list the key assumptions and parameter values wherein the PA runs differed, so that the reasons for the large differences in dose could be better understood.

The first column of the table simply provides an item number to simplify this discussion. Run numbers are also given for the runs made in this study. The next four columns indicate the waste evaluated, the inventory of the key dose contributors, the conceptual location of the disposal, and the FWF studied. The next two columns list the assumptions made for the FWF release process and the assumed K_d values for the key dose contributors. The groundwater travel time in the vadose zone is listed next, followed by the calculated maximum value of the groundwater dose.

Table 10-24. Comparison of assumptions and results of PA runs involving Sites D and E.

Item (run no.)	Waste	Inventory of key dose contributors	Location of disposal	Final waste form	Assumptions concerning release process	Assumed values for K_d s	Groundwater travel time in vadose zone (yr)	Peak groundwater dose (mrem/yr) ^d
1	Hanford LLW	5.23 Ci C-14 0.22 Ci I-129 1.69 Ci Tc-99 31 Ci Uranium (nuclide not specified)	Hanford	Cement	Realistic—Release rate limited by solubility of elements in cement pore water	C, I, Tc 0; Am 100 U 0	450	0.47
2 (Run 14)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	Hanford	Cement	Conservative—no chemical adsorption in cement	Am 100 U 0	450	3400
3 (Run 16)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	Hanford	IEB4	Realistic—dissolution	Am 100 U 0	450	20
4	SRS LLW	back-calculated: Am-241 1000 Np-237 0.2 I-129 5.3E-4 Cs-135 19 Sr-90 3E13 etc.	SRS	Cement	Realistic—Release rate limited by solubility of elements in cement pore water	Am 1900/150 ^a Np 5/10 U 35/50 I 1.0/0.6 Sr 15/10 Cs 280/100	4.52	25, by constraint
5 (Run 13)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	SRS	Cement	Conservative—no chemical adsorption in cement	Am 1900/150 U 35/50	4.52	1100
6 (Run 15)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	SRS	IEB4	Realistic—dissolution	Am 1900/150 U 35/50	4.52	44
7 (Run 1)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	INEL SDA	Cement	Conservative—no chemical adsorption in cement	Am 700/70 ^b U 1000/100	933	3.1
8 (Run 1IR2NB)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	INEL SDA	Cement	Conservative—no chemical adsorption in cement	Am 700/70 U 1000/100	6.3 ^c	7.7

Table 10-24. (continued).

Item (run no.)	Waste	Inventory of key dose contributors	Location of disposal	Final waste form	Assumptions concerning release process	Assumed values for K_d s	Groundwater travel time in vadose zone (yr)	Peak groundwater dose (mrem/yr) ^d
9 (Run 1INT)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	INEL SDA	Cement	Realistic—chemical adsorption	Am 700/70 U 1000/100	933	0.025
10 (Run 9)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	INEL Site 14	Cement	Conservative—no chemical adsorption in cement	Am 700/70 U 1000/100	400	0.78
11 (Run 9INT)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	INEL Site 14	Cement	Realistic—chemical adsorption	Am 700/70 U 1000/100	400	0.0066
12 (Run 3)	INEL ALLW	1340 Ci Am-241 2401 Ci Pu-241 13 Ci U-233	INEL SDA	IEB4	Realistic—dissolution	Am 700/70 U 1000/100	933	0.011

a. First value is for sand; second value is for soil.

b. First value is for sediments; second value is for basalt.

c. No credit was taken for the basalt in the vadose zone. In addition, the infiltration rate was increased from 1.32 cm/yr to 10 cm/yr.

d. Performance objective is 25 mrem/yr.

The following observations were made from the entries in Table 10-24:

- The inventories in the SRS waste and Hanford waste differed somewhat from that used in this study. The INEL ALLW tends to be dominated by the Am-241, Pu-239, and U-233 decay chains. The Hanford waste was dominated by U, the specific isotopic composition of which is not known to the authors. The SRS inventory, which had been back-calculated to derive limits for various radionuclides, tended to be dominated by Am-241, Np-237, and certain nonactinides.
- The K_d values used at Site D were generally similar to those used for Sites A, B, and C, although much lower for U. The values at Site E were considerably lower than those at the other sites, especially for U. The extent to which the differences reflect different geochemical interactions at the various sites versus differences in conservatism of the assumptions is not known.
- At the SDA, increasing the assumed infiltration rate to the maximum and taking no credit for the underlying basalt layers increases the dose by only about a factor of 2 1/2, in spite of decreasing the vadose zone travel time from 933 yr to 6.3 yr (see items 7 and 8).
- At either the SDA or Site 14, including the chemical adsorption in the cement lowers the dose by about 2 orders of magnitude (see items 7 and 9, and items 10 and 11, respectively). In fact, including the chemical adsorption reduces the dose for cement to only a factor of about 3 larger than that for IEB4 (see items 9 and 12).
- For the INEL ALLW in a cement FWF with no chemical adsorption, the dose at SRS is about 350 times higher than that at the SDA and fails the performance objective by a very large margin (see items 5 and 7). Even if the INEL vadose zone travel time were reduced from 933 yr to 6.3 yr, close to the 4.52 yr at SRS, the dose at SRS is still about 140 times higher than at the SDA (see items 5 and 8). Evidently, the much lower value of K_d for U (35 in SRS sediments versus 1,000 in INEL sediments, a difference of a factor of almost 30) is the controlling parameter.
- For the INEL ALLW at SRS, even an IEB4 FWF fails the performance objective (see item 6). Thus, with these assumptions, no FWF would pass the performance objective when disposing of the INEL ALLW at SRS.
- For the INEL ALLW in a cement FWF with no chemical adsorption, the dose at Hanford is about 1,100 times higher than that at the SDA and fails the performance objective by a very large margin (see items 2 and 7). Even if the INEL vadose zone travel time were reduced from 933 yr to 6.3 yr (Hanford's is 450 yr), the dose at Hanford is still about 440 times higher than at the SDA (see items 2 and 8). Evidently, the much lower value of K_d for U (0 at Hanford versus 1,000 in INEL sediments) is the controlling factor.
- For the INEL ALLW in an IEB4 FWF at Hanford, the dose barely satisfies the performance objective (see item 3).

10.2.18.3 Results of Additional Runs. Based on the above observations, additional runs were made to (a) substantiate that the differences are actual and not due to a calculational error and (b) explore briefly the reasons for the large differences. Detailed evaluation of the reasons for the differences is beyond the scope of this study.

The following conclusions were drawn from the results of the new runs, which are presented in Table 10-25. The first three conclusions involve the PCC FWF. The fourth conclusion involves the IEB4 FWF.

- Taking credit for the chemical adsorption of INEL ALLW radionuclides in a PCC FWF at Site D or E reduces the dose by about 2 orders of magnitude. This can be seen by comparing the results of Run 14INT against item 2 (Run 14) in Table 10-24, and by comparing Run 13INT against item 5 (Run 13) in Table 10-24. This fractional dose reduction of about 2 orders of magnitude is the same as that found for disposal of PCC at Sites A and B in Section 10.2.13. Thus, taking credit for the chemical adsorption in the PCC accounts for much of the large difference in dose at Sites D and E. The PAs for Hanford and the SRS took credit for the chemical interactions between the radionuclides and the cement, although a detailed analysis of how the chemical interactions were modeled at those sites is beyond the scope of these comparisons.
- For purposes simply of illustrating the sensitivity of dose to K_d values, use of the less conservative values of the K_d s—the INEL values—for the migration of INEL ALLW radionuclides in PCC at Site D or E reduces the dose by about 3 orders of magnitude at Site E and 1 order of magnitude at Site D. This can be seen by comparing the results of Run 14INELKD against item 2 (Run 14) in Table 10-24, and by comparing Run 13INELKD against item 5 (Run 13) in Table 10-24. Thus, the difference in K_d values explains much of the difference in dose.
- Combining the effects of chemical adsorption of radionuclides in the PCC and the use of different values of K_d s reduces the dose even more. The dose is reduced by about 4 orders of magnitude at Site E and 2 to 3 orders of magnitude at Site D. This can be seen by comparing the results from Run 14INT/INELKD against item 2 (Run 14) in Table 10-24, and by comparing Run 13INT/INELKD against item 5 (Run 13) in Table 10-24.
- Using less conservative values of the K_d s—the INEL values—for the migration of INEL ALLW radionuclides in IEB4 at Site D or E has a much smaller effect than was the case for PCC. With IEB4, the dose reduction is only a factor of about 70 at Site E and a factor of about 8 at Site D. This can be seen by comparing the results of Run 16INELKD against item 3 (Run 16) in Table 10-24, and by comparing Run 15INELKD against item 6 (Run 15) in Table 10-24. The reduced effect in this case is because the IEB4 FWF has a larger effect than does PCC in reducing the release rate.

Table 10-25. New runs to help clarify the large differences in PA runs for INEL ALLW disposed of at Sites D and E. (All runs involve INEL ALLW. Because of the complex, focused nature of the set of runs listed here, they are not included in Table 10-2.)

Run No.	Item in Table 10-24 to which results are compared	Location of disposal	Final waste form	Assumptions concerning release process	Assumed values for K_d s	Groundwater travel time in vadose zone (yr)	Maximum dose (mrem/yr)	Time of maximum dose (yr)
14INT	2 (Run 14)	Hanford	PCC	Realistic—chemical adsorption in cement	Same as Run 14 (standard values for Hanford in this study)	450	18	470
13INT	5 (Run 13)	SRS	PCC	Realistic—chemical adsorption in cement	Same as Run 13 (standard values for SRS in this study)	4.52	26	10,000
14INELKD	2 (Run 14)	Hanford	PCC	Conservative—no chemical adsorption in cement	Use INEL K_d s	450	3.7	53,000,000
14INT/ INELKD	2 (Run 14)	Hanford	PCC	Realistic—chemical adsorption in cement	Use INEL K_d s	450	0.14	53,000,000
16INELKD	3 (Run 16)	Hanford	IEB4	Realistic—dissolution	Use INEL K_d s	450	0.29	53,000,000
13INELKD	5 (Run 13)	SRS	PCC	Conservative—no chemical adsorption in cement	Use INEL K_d s	4.52	53	120,000
13INT/ INELKD	5 (Run 13)	SRS	PCC	Realistic—chemical adsorption in cement	Use INEL K_d s	4.52	2.6	130,000
15INELKD	6 (Run 15)	SRS	IEB4	Realistic—dissolution	Use INEL K_d s	4.52	5.6	120,000

10.2.18.4 Overall Conclusions. The doses calculated herein for the disposal of INEL ALLW at Sites D and E are very large on an absolute scale. They are also very large compared with the corresponding doses calculated for (a) disposal of INEL ALLW at Sites A, B, and C and for (b) disposal of the waste from Sites D and E at their respective sites. The dose differences appear to be caused primarily by the following differences in assumptions and parameter values:

- Different inventories for the waste from Sites D and E compared with the INEL ALLW
- Taking no credit for chemical adsorption of radionuclides in the cement FWF for most of the runs herein.
- Use of smaller values of K_d at Sites D and E
- The shorter vadose zone travel time of the groundwater at Site D.

As stated in Section 10.2.18.1, additional factors may have affected the calculations, particularly for Sites D and E. The above conclusions are based only on the factors investigated herein.

10.2.19 Compliance with Performance Objective to Protect Groundwater Resources

Section 1.4.1 listed the performance objectives to be complied with by the waste disposal system. One performance objective involves the protection of groundwater resources, consistent with federal, state, and local requirements. As shown in Tables 1-1 and 10-1, that performance objective was interpreted in this study to entail compliance with four EPA limits in 40 CFR 141 for radionuclides in community drinking water systems.

The study predicted the extent of compliance with the four EPA limits. Runs covering the original assumptions for the various FWFs at the various disposal sites were examined. Tables 10-26a through 10-26d present the results, which can be summarized, respectively, as follows:

- Beta- and photon-emitters dose limit: For Sites A, B, and C, all FWFs pass. For Sites D and E, all FWFs except IEB4 fail.
- Alpha-emitters concentration limit: For Sites A, B, and C, all FWFs pass. For Sites D and E, all FWFs fail.
- Ra-226 and Ra-228 concentration limit: All FWFs pass at all sites.
- U concentration limit: For Sites A, B, and C, all FWFs pass. For Sites D and E, all FWFs except IEB4 fail.

The composite conclusion regarding compliance with these four limits is that all FWFs pass at Sites A, B, and C, but all FWFs fail at Sites D and E.

Table 10-26. Compliance with performance objective to protect groundwater resources.

(a) Compliance with the objective to limit the dose from beta-emitters and photon-emitters in groundwater to 4 mrem/yr.

Runs with doses below the regulation

Run	Site	FWF	Dose from beta-emitters and photon-emitters (mrem/yr)
1	A	PCC	4.3E-10
2	C	PCC	3.2E-13
3	A	IEB4	1.5E-12
4	C	IEB4	4.8E-14
6	B	IEB	2.0E-12
8	B	IEB4	3.3E-13
9	B	PCC	1.1E-10
10	B	FUETAP	7.2E-11
11	B	BSG	1.1E-11
12	B	ASG	2.0E-12
15	D	IEB4	3.0E+00
16	E	IEB4	1.3E+00
17	A	ASG	9.0E-12
18	A	IEB	9.2E-12
1UTR	A	UTR	4.9E-10
9UTR	B	UTR	1.3E-10

Runs with doses that exceed the regulation

Run	Site	FWF	Dose from beta-emitters and photon-emitters (mrem/yr)
13	D	PCC	7.6E+01
14	E	PCC	1.9E+02
19	E	IEB	7.7E+00
13UTR	D	UTR	5.4E+01
14UTR	E	UTR	3.9E+02

Table 10-26. (continued).

(b) Compliance with the objective to limit the total concentration of alpha-emitters to 15 pCi/L (including Ra-226, excluding U and Rn).

Runs with concentrations below the regulation

Run	Site	FWF	Total concentration of alpha-emitters (pCi/L)
1	A	PCC	6.5E-01
2	C	PCC	4.8E-04
3	A	IEB4	2.3E-03
4	C	IEB4	7.2E-05
6	B	IEB	3.1E-03
8	B	IEB4	5.0E-04
9	B	PCC	1.6E-01
10	B	FUETAP	1.1E-01
11	B	BSG	1.7E-02
12	B	ASG	3.1E-03
17	A	ASG	1.4E-02
18	A	IEB	1.4E-02
1UTR	A	UTR	7.4E-01
9UTR	B	UTR	1.9E-01

Runs with concentrations that exceed the regulation

Run	Site	FWF	Total concentration of alpha-emitters (pCi/L)
13	D	PCC	1.1E+03
14	E	PCC	2.7E+03
15	D	IEB4	4.3E+01
16	E	IEB4	1.9E+01
19	E	IEB	1.1E+02
13UTR	D	UTR	7.7E+02
14UTR	E	UTR	5.6E+03

Table 10-26. (continued).

(c) Compliance with the objective to limit the concentration of Ra-226 and Ra-228 combined to 5 pCi/L.

All runs predict concentrations below the regulation

Run	Site	FWF	Combined concentration of Ra-226 and Ra-228 (pCi/L)
1	A	PCC	4.6E-06
2	C	PCC	<1E-10
3	A	IEB4	3.6E-08
4	C	IEB4	<1E-10
6	B	IEB	3.8E-07
8	B	IEB4	6.3E-08
9	B	PCC	9.3E-06
20	B	FUETAP	7.0E-06
11	B	BSG	2.0E-07
12	B	ASG	3.8E-06
13	D	PCC	2.0E+00
14	E	PCC	6.8E-02
15	D	IEB4	4.2E-02
16	E	IEB4	9.6E-05
17	A	ASG	2.1E-07
18	A	IEB	2.2E-07
19	E	IEB	1.9E-03
1UTR	A	UTR	1.9E-05
9UTR	B	UTR	3.3E-05
13UTR	D	UTR	1.5E+00
14UTR	E	UTR	4.8E-02

Table 10-26. (continued).**(d) Compliance with the objective to limit the concentration of uranium to 30 pCi/L.**Runs with concentrations below the regulation

Run	Site	FWF	Concentration of uranium (pCi/L)
1	A	PCC	2.2E-01
2	C	PCC	1.7E-04
3	A	IEB4	7.7E-04
4	C	IEB4	2.5E-05
6	B	IEB	1.1E-03
8	B	IEB4	1.7E-04
9	B	PCC	5.6E-02
10	B	FUETAP	3.7E-02
11	B	BSG	5.8E-03
12	B	ASG	1.1E-03
15	D	IEB4	2.7E+01
16	E	IEB4	1.6E+01
17	A	ASG	4.7E-03
18	A	IEB	4.8E-03
1UTR	A	UTR	2.5E-01
9UTR	B	UTR	6.6E-02

Runs with concentrations that exceed the regulation

Run	Site	FWF	Concentration of uranium (pCi/L)
13	D	PCC	9.2E+02
14	E	PCC	3.1E+04
19	E	IEB	9.0E+01
13UTR	D	UTR	6.4E+02
14UTR	E	UTR	6.5E+04

10.3 Results for Undisturbed-Waste Scenario: Atmospheric Pathway

The methodology described in Section 8 was used to estimate the radon flux and the concentration at and dose to a receptor on the downwind edge of the disposal unit. A PCC FWF was assumed.

Two cases were considered: (a) the dose to a receptor just before 500 yr, and (b) the dose to a receptor at 167,000 yr. The first case included the presence of 16 ft of soil (11 ft of abovegrade cover and 5 ft of belowgrade backfill). The second case corresponded to the peak Rn-222 production rate from the inventory and was modeled with only the 5 ft of belowgrade backfill remaining.

The results are shown in Table 10-27. For the first case, the radon flux from the ground surface above the disposed waste was calculated to be $1.9\text{E-}05$ pCi/m²-sec. For the second case, the flux was estimated to be 0.84 pCi/m²-sec.

No specific regulations exist for the control of radon emissions (radon flux) from a LLW disposal facility. The EPA has identified an emission limit of 20 pCi/m²-sec as a basis for remedial actions involving mill tailings for the Uranium Mill Tailings Remedial Action (UMTRA) project (DOE 1989). The two values calculated here are well below that limit.

Given a value of $1.9\text{E-}05$ pCi/m²-sec for the 500-yr radon flux, the Eulerian box model yielded a value of $2.4\text{E-}07$ pCi/L for the concentration of radon at the receptor, located outdoors. The resulting dose is $4.8\text{E-}05$ mrem/yr.

For the 167,000-yr case, the Eulerian box model yielded a value of $1.1\text{E-}02$ pCi/L for the concentration of radon at the receptor. The resulting dose is 2.1 mrem/yr.

There are also no specific standards for the outdoor concentration of radon and the corresponding dose. The average concentration of radon in homes in the United States, resulting from naturally occurring radon, is about 1 pCi/L. The resulting equivalent whole body dose is 200 mrem/yr (Lodge 1991). The EPA has established an action level for indoor radon of 4 pCi/L, above which remedial action is recommended (EPA 1987). The EPA estimates the average outdoor radon level to be 0.4 pCi/L (Lodge 1991). Thus, the outdoor concentrations calculated here are less than one-tenth of the natural background concentrations of radon in outdoor air.

Table 10-27. Dose from radon release in undisturbed-waste scenario.

Yr since disposal	Concentration of radon in FWF (pCi/L)	Total soil cover (ft)	Flux from ground surface (pCi/m ² -sec)	Concentration at receptor (pCi/L)	Dose rate to receptor (mrem/yr)
Just less than 500	6.0	16	$1.9\text{E-}05$	$2.4\text{E-}07$	$4.8\text{E-}05$
167,000	12,000	5	0.84	$1.1\text{E-}02$	2.1

This evaluation was conservative for the following reasons:

- Leaching of the radon parents and migration to greater depths below the surface was neglected
- The most conservative values were used as inputs for the Eulerian box model, which resulted in the highest concentration of radon at the receptor.

10.4 Results for Intrusion Scenarios

Section 10.4 provides the results of the cases evaluated for the four intrusion scenarios. Section 10.4.1 provides an introduction and the rationale for the cases evaluated. Sections 10.4.2 through 10.4.11 discuss the effects on the results for the first three intrusion scenarios when several parameters and assumptions were changed. Section 10.4.12 provides the results for the fourth scenario, intruder-radon.

10.4.1 Introduction and Rationale

10.4.1.1 General. To within the precision of this study, the results for the intrusion scenarios are independent of the location of the waste disposal site.

The intrusion results depend strongly upon the burial depth (belowgrade) and on the presence or absence of the abovegrade multi-layer, engineered cover. The latter depends, in turn, on the time at which the intrusion is assumed to occur. The reader is referred to Section 5.2 concerning the layout of the waste disposal pit and the abovegrade cover and to Section 4.3 concerning time periods.

For the intrusion scenarios, two different burial depths were evaluated at two different intrusion times, although not all of the combinations of these factors were evaluated in detail. One burial depth (termed for simplicity the "16-ft cover") initially has 5 ft of backfill soil belowgrade plus an 11-ft abovegrade engineered cover, totaling 16 ft of material overlying the waste. The other burial depth (termed the "20-ft cover") initially has 9 ft of backfill soil belowgrade plus the same 11-ft abovegrade engineered cover, totaling 20 ft of material overlying the waste.

As discussed in Section 5.2, the cover was assumed to remain intact for all times up to but not including 500 yr postclosure. At 500 yr and later, no credit was taken for the abovegrade cover. The belowgrade backfill was assumed to be present at all times.

The total thickness of the soil cover affects the doses from a drilling event by varying the volume of clean soil within the drill hole that is mixed with the waste. The cover thickness also affects doses from an excavation event by varying the volume of waste (if any) that is excavated. By virtue of its effects on the drilling and excavation events, the total thickness of the cover affects the doses in the intruder-agricultural scenario. Finally, the thickness affects the doses in the intruder-radon scenario.

Two different intrusion times were evaluated: 100 yr and 500 yr following closure of the disposal facility. The rationale for evaluating the performance at these times is discussed briefly in Section 4.3 and in detail in Sections 10.4.1.2 through 10.4.1.5.

Confirming runs were performed for numerous other times up through 100,000 yr and are reported in Section 10.4.10. The results confirm that the arguments made below concerning the times of peak dose are valid.

The intruder-radon scenario was a special case. The doses were evaluated at the time of maximum radon concentration. These times were 4,600 yr for PCC and 20,000 yr for IEB4.

The burial depths and scenario times evaluated for the intrusion scenarios are shown in Table 10-28. The detailed rationale for the selected combinations follows in Sections 10.4.1.2 through 10.4.1.5.

10.4.1.2 Rationale for Cases Evaluated in Intruder-Drilling Scenario. The intruder-drilling scenario was evaluated at 100 yr with both the 16- and 20-ft soil covers (see the "D" entries in Table 10-28).

Evaluations of the intruder-drilling scenario at 500 yr were not performed because these would be expected to yield lower doses, based on the following rationale. Two factors affect the intruder-drilling dose at 500 yr compared with the dose at 100 yr. The first factor is radioactive decay of the radionuclides that contribute most of the dose in this scenario. This first factor tends to decrease the dose as the time increases. Typically, 90% to 93% of the dose in the intruder-drilling scenarios at 100 yr was found to be contributed by Cs-137. The remaining 7% to 10% was contributed by Am-241. At 500 yr, the Cs-137 inventory (half-life of about 30 yr) would have decayed by a factor on the order of 10,000, so its contribution to dose would be negligible compared with that of the Am-241. The Am-241 is relatively long-lived (half-life of about 433 yr) and its first decay product, Np-237, is extremely long-lived. Thus, at 500 yr, the Am-241 would constitute most of the dose. Based on considering this factor only, the dose at 500 yr would be expected to be less than one-tenth of the dose at 100 yr.

The second factor involves the loss of the abovegrade cover just before 500 yr. This loss results in decreasing the dilution factor for the drill cuttings by about one third, tending to increase proportionately the dose at 500 yr compared to the dose at 100 yr.

The first factor predominates over the smaller second factor, so the intruder-drilling dose at 500 yr is expected to be less than the dose at 100 yr.

10.4.1.3 Rationale for Cases Evaluated in Intruder-Excavation Scenario. The intruder-excavation scenario was evaluated only at 500 yr and only with the 16-ft cover (the "E" entry in Table 10-28) for the following reasons.

Table 10-28. Burial depths and intrusion times evaluated for the intrusion scenarios.

Design\time	100 yr	500 yr	At time of calculated peak dose
16-ft soil cover	D	E, A	—
20-ft soil cover	D, A	A (limited)	R

D = intruder-drilling scenario.

E = intruder-excavation scenario.

A = intruder-agricultural scenario.

R = intruder-radon scenario.

Limited = only a few FWFs analyzed.

In the case of the 16-ft cover, that cover will be present at facility closure and will be maintained during the 100-yr period of institutional control. Excavation of an 8-ft basement in this cover material at 100 yr will not disturb the waste and will therefore result in a minuscule acute dose. The 11-ft abovegrade cover overlying the 5 ft of belowgrade soil has a 500-yr design life, however. At the end of the design life, the abovegrade cover is assumed to disappear, leaving only the 5 ft of belowgrade backfill soil. Excavation of a basement 8 ft deep in this remaining material at 500 yr was assumed to disturb a layer of waste 3 ft thick (8-ft basement - 5-ft backfill = 3 ft of waste penetrated).

In the case of the 20-ft cover, an intruder-excavation scenario at 500 yr would result in a smaller acute dose, for the following reason. At 500 yr, no credit is taken for the abovegrade cover, so the 9 ft of belowgrade backfill remains. The 8-ft basement would not penetrate completely through the backfill, so no waste would be disturbed. Therefore, this case was not evaluated for the excavation scenario.

10.4.1.4 Rationale for Cases Evaluated in Intruder-Agricultural Scenario. The rationale for the cases evaluated in the intruder-agricultural scenario basically follows from the preceding rationale for the drilling and excavation scenarios. Those scenarios are the initiating events for the agricultural scenario.

- For the 16-ft cover, the intruder-agricultural scenario dose at 500 yr was evaluated because it is expected to be larger than the dose at 100 yr. At 100 yr, the abovegrade cover is intact and only the drilling event penetrates the waste (excavation does not). At 500 yr, no credit is taken for the abovegrade cover, so excavation penetrates the waste and a relatively large volume is exhumed. (Excavation exhumes about 200 times the volume of waste exhumed by drilling.) This much larger exhumation leads to an increased dose at 500 yr, compared with that at 100 yr. (The key dose contributors are

the relatively long-lived Am-241 and Pu-239, so radioactive decay does not play a major role between 100 and 500 yr.) Thus, for the 16-ft cover, intrusion at 500 yr was expected to result in the largest dose for the agricultural scenario and was evaluated.

- For the 20-ft cover, excavation does not penetrate the waste at either 100 yr or 500 yr. Drilling was assumed to be the initiating event. As discussed in Section 10.4.1.2, the radioactive decay in the key radionuclides between 100 yr and 500 yr is expected to reduce the dose at 500 yr more than the loss of the abovegrade cover increases the dose (due to the reduced dilution of the cuttings). Therefore, the dose at an intrusion time of 100 yr is expected to result in the largest agricultural dose for the 20-ft cover and was the dose evaluated.

The "A" entries in Table 10-28 show that the intruder-agricultural scenario was evaluated for the 16-ft cover at 500 yr and for the 20-ft cover at 100 yr. In addition, a limited number of cases were evaluated for the 20-ft cover at 500 yr.

10.4.1.5 Summary of Cases Evaluated. Table 10-29 shows the different intrusion combinations (or cases) and which combinations were evaluated (an "X" means that the combination was evaluated). Not all combinations were evaluated, but those that were illustrate the effects of the various factors on the results.

10.4.2 Effects of FWF on Results

Table 10-30 lists the results by FWF for the cases evaluated for the three intrusion scenarios (drilling, excavation, and agricultural).

For both cases of the intruder-drilling scenario and for the intruder-excavation scenario, all FWFs satisfy the performance objective of 500 mrem acute exposure.

For the agricultural scenario with the 16-ft cover, all FWFs fail at 500 yr. With the 20-ft cover, all FWFs comply at 100 yr except for IEB and IEB4, which are slightly above the performance objective of 100 mrem/yr chronic exposure. ASG is only slightly under the limit.

The effects of FWF density and waste loading on the intrusion doses are evident from Table 10-30. The more concentrated, advanced FWFs exhibit the higher doses. This is because the product of the waste loading, FWF density, and volume of excavated waste equals the mass of waste brought to the surface.

Another key parameter affecting dose that was not varied here is related to facility design. These evaluations assumed a subsurface waste stack of eight 55-gal drums (22.7 ft) and total cover thicknesses of 16 or 20 ft. Reducing the stack height would proportionally reduce the predicted intruder doses from drilling events.

Table 10-29. Cases evaluated for the intrusion scenarios.

FWF	Intruder-drilling 16-ft soil cover ^a at 100 yr	Intruder-drilling 20-ft soil cover ^b at 100 yr	Intruder-excavation 16-ft soil cover ^a at 500 yr	Intruder-agricultural 16-ft soil cover ^a at 500 yr	Intruder- agricultural 20-ft soil cover ^b at 100 yr	Intruder-agricultural 20-ft soil cover ^b at 500 yr
PCC	X	X	X	X	X	—
FUETAP	X	X	X	X	X	—
BSG	X	X	X	X	X	—
ASG	X	X	X	X	X	—
IEB4	X	X	X	X	X	X
IEB(10) ^c	X	X	X	X	X	—
IEB(100) ^c	X	X	X	X	X	X
IEB(1000) ^c	X	X	X	X	X	—
Untreated waste	X	X	X	X	X	—

a. Here and in subsequent tables, the 16-ft soil cover refers to disposal with 5 ft of belowgrade backfill and an 11-ft abovegrade multilayer cover of 500-yr design life.

b. Here and in subsequent tables, the 20-ft soil cover refers to disposal with 9 ft of belowgrade backfill and an 11-ft abovegrade multilayer cover of 500-yr design life.

c. Value in parentheses is the TRU activity concentration in nCi/g. All other FWFs had concentrations of 100 nCi/g.

Table 10-30. Effects of FWF on performance in intrusion scenarios.

FWF	FWF density (g/cm ³)	FWF waste loading (wt. %)	Intruder-drilling 16-ft soil cover at 100 yr (mrem)	Intruder-drilling 20-ft soil cover at 100 yr (mrem)	Intruder-excavation 16-ft soil cover at 500 yr (mrem)	Intruder-agricultural 20-ft soil cover at 100 yr (mrem/yr)	Intruder-agricultural 16-ft cover at 500 yr (mrem/yr)
PCC	1.95	40	23	18	96	46	1,400
FUETAP	1.95	40	23	18	96	46	1,400
BSG	2.65	30	23	18	96	46	1,400
ASG	2.65	60	46	36	190	92	2,700
IEB	3.00	60	49	42	220	110	3,100
IEB4	3.50	60	59	49	260	120	3,600

10.4.3 Effects of TRU Radionuclide Activity Concentration on Results

Table 10-31 lists the results of the various cases evaluated for the intrusion scenarios wherein the concentration of TRU radionuclide activity in the IEB FWF was varied.

The results are nonlinear, due to combined contributions from actinides and nonactinides. For example, the intruder-drilling runs for a 20-ft soil cover at 100 yr yield acute doses of 39, 42, and 72 mrem for the three concentrations of 10, 100, and 1,000 nCi/g TRU activity, respectively. The dominant radionuclide contributing to the dose is Cs-137, which gives a constant dose of about 39 mrem to the lungs from external exposure in all three cases. Added to this dose is a contribution from Am-241 (and to a lesser extent Pu-239), which provides additional doses of 0.31, 3.1, and 31 mrem, respectively, to bone surfaces from inhalation. These results are a consequence of the inventory assumptions, which held the nonactinide inventory constant while varying the actinide inventory over 2 orders of magnitude.

For the intruder excavation scenario (16-ft soil cover, 500 yr), the doses are controlled by the inhalation route, with Pu-239, Pu-240, and Am-241 as the dominant contributors. As expected, the dose increases linearly with TRU concentration, by virtue of the inventory assumptions.

For the intruder-agricultural scenario, only one case of which (20-ft soil cover, 100 yr) is shown in Table 10-31, the chronic doses were estimated at 47, 110, and 690 mrem/yr for the three TRU radionuclide concentrations. Cs-137 provides a dose of 24 mrem/yr for all three runs. Sr-90 provides doses of about 15 mrem/yr in each of the three cases. In addition, Am-241 contributes 4.3, 43, and 430 mrem/yr for the three cases. Pu-239 and Pu-238 also contribute 110 and 76 mrem/yr, respectively, for the TRU concentration of 1,000 nCi/g and proportionately smaller doses for the smaller concentrations. Other radionuclides contribute lesser portions of the dose. Table 10-3 includes the results for the other case evaluated (16-ft soil cover, 500 yr).

Table 10-31. Effects of TRU activity concentration in IEB on performance in intrusion scenarios.

TRU activity (nCi/g)	Intruder-drilling 16-ft soil cover 100 yr (mrem)	Intruder-drilling 20-ft soil cover 100 yr (mrem)	Intruder- excavation 16-ft soil cover 500 yr (mrem)	Intruder- agricultural 20-ft soil cover 100 yr (mrem/yr)
10	45	39	22	47
100	49	42	220	110
1,000	84	72	2200	690

10.4.4 Effects of Disposal Depth on Results

Table 10-32 indicates the cases evaluated to investigate the effects of burial depth. As stated previously, two burial depths were evaluated. (Burial at a depth requiring 5 ft of belowgrade backfill results in a 16-ft cover, including the 11-ft abovegrade cover. Burial at a depth requiring 9 ft of belowgrade backfill results in a 20-ft cover.)

A comparison of the intruder-drilling scenario results at 100 yr, assuming 16 ft versus 20 ft of total cover, shows an increase in acute dose of approximately 25% with the 16-ft cover. This increased dose occurs because a smaller volume of clean material in the drill hole is mixed with the exhumed waste. In all cases, the predicted doses from the intruder-drilling scenarios are less than the performance objective limit of 500 mrem.

The effect of burial depth on the intruder-agricultural dose at 500 yr was investigated briefly. The doses from the shallower burial (16-ft cover) exceeded the 100 mrem/yr limit for all FWFs. However, with the deeper burial (20-ft cover), the doses were dramatically reduced. The dose for IEB4 (the FWF with the largest dose) was reduced from 3,600 mrem/yr to 25 mrem/yr. For IEB, the dose was reduced from 3,100 to 22 mrem/yr. Thus, all FWFs meet the 100 mrem/yr performance objective at 500 yr if the deeper burial is implemented.

The explanation for the large reduction in agricultural dose when the burial depth is increased is as follows. With only 5 ft of belowgrade backfill after the abovegrade cover is gone, an 8-ft excavation exhumes a relatively large amount of waste. With 9 ft of belowgrade backfill, the excavation does not penetrate the waste. Only the drilling penetrates the waste, and the volume exhumed is much smaller.

10.4.5 Effects of Untreated Waste on Results

The results for the intrusion scenarios for untreated waste are shown in Table 10-33 (see Section 2.4.5 for the radionuclide inventory). The results for a PCC FWF are listed for comparison.

In these evaluations, the untreated waste was distributed uniformly throughout the disposal facility. Thus, each scenario exhumes equal fractions of the different types of untreated waste.

The doses in all of the intrusion scenarios are lower for the untreated waste than for PCC. The relative percentage of each radionuclide's contribution to dose is the same for both waste forms. The total radionuclide inventory is held constant. Since different treated and untreated waste forms have different densities and mass loadings of the waste, the radionuclide concentration also differs for each waste form. Intruder scenario doses are proportional to these radionuclide concentrations.

For the 100-yr intruder-drilling scenario with a 16-ft cover, the dose from untreated waste is 12 mrem, roughly half of the dose from PCC (23 mrem). The dominant dose contribution (92%) is from Cs-137 and is a lung dose from external radiation.

Table 10-32. Effects of burial depth on performance in intrusion scenarios.

FWF	Intruder-drilling 16-ft soil cover at 100 yr (mrem)	Intruder-drilling 20-ft soil cover at 100 yr (mrem)	Intruder- agricultural 16-ft soil cover at 500 yr (mrem/yr)	Intruder- agricultural 20-ft soil cover at 500 yr (mrem/yr)
PCC	23	18	1,400	—
FUETAP	23	18	1,400	—
BSG	23	18	1,400	—
ASG	46	36	2,700	—
IEB	49	42	3,100	22
IEB4	59	49	3,600	25

Table 10-33. Intrusion scenario results for untreated waste.

Scenario	Dose from untreated waste	Dose from PCC	Radionuclides contributing $\geq 5\%$ of dose	Radionuclide contribution (%)
Intruder-drilling 16-ft cover, 100 yr	12 mrem	23 mrem	Cs-137 Am-241	92 8
Intruder-excavation 16-ft cover, 500 yr	58 mrem	96 mrem	Pu-239 Am-241 Pu-240	50 36 12
Intruder-agricultural 16-ft cover, 500 yr	820 mrem/yr	1,400 mrem/yr	Am-241 Pu-239 Pu-240	60 32 7
Intruder-drilling 20-ft cover, 100 yr	9.6 mrem	18 mrem	Cs-137 Am-241	90 9
Intruder-agricultural 20-ft cover, 100 yr	24 mrem	46 mrem	Am-241 Sr-90 Cs-137 Pu-239 Pu-238	46 14 13 13 9

The total volume of untreated waste is 24,390 m³. The total volume of the same inventory treated and encapsulated as a PCC waste form is 15,300 m³. From consideration of these volumes, one would expect the doses from untreated waste to be 63% rather than 48% of those from PCC. Revisions in the refined inventory of Cs-137 account for the difference (see Appendices A and B). The PCC analyses assumed a Cs-137 inventory of 1,125 Ci. This assumption was refined to a value of 936.3 Ci before performing the analysis of untreated waste. When the difference in total waste volume and the revision in Cs-137 inventory are taken into account, the results for untreated waste and PCC are consistent. Predicted doses for both waste forms are below the performance objective of 500 mrem for acute exposures.

For the 500-yr intruder-excavation scenario with a 16-ft cover, the dose from untreated waste is 58 mrem, which is less than the dose from PCC (96 mrem). The dominant dose contribution (50%) is from Pu-239 delivered to lungs via inhalation. Other principal contributors are Am-241 and Pu-240, which provide doses through the same exposure route. Predicted doses for both waste forms are below the performance objective of 500 mrem for acute exposures.

For the 500-yr intruder-agricultural scenario with a 16-ft cover, the dose from untreated waste is 820 mrem/yr, which is less than the dose from PCC (1,400 mrem/yr). The dominant dose contribution (60%) is from Am-241 delivered to bone surfaces from inhalation. Other contributors are Pu-239 and Pu-240, which provide doses through the same exposure route.

The other cases listed in Table 10-33 illustrate the same relationship between the doses from untreated waste and those from the PCC FWF.

10.4.6 Effects of Refined Inventory on Results

Section 2.4.3 describes the refined radionuclide inventory, in which the CH waste and RH waste were modeled separately instead of being commingled as was the case in all the intrusion runs discussed thus far. Evaluations were performed using this refined inventory for the drilling, excavation, and agricultural intrusion scenarios.

These evaluations were identical to those performed previously, except that separate evaluations were performed for the CH and RH fractions of the FWF. Doses were calculated assuming an IEB or a PCC FWF.

Table 10-34 provides separate results for the CH, RH, and combined inventory fractions.

The doses from the intruder-drilling scenario with 16-ft soil cover at 100 yr are dominated by Cs-137, except in the CH cases. The majority (94%) of the Cs-137 in the refined inventory is in the RH fraction, so direct interception of RH waste in a drilling scenario yields extremely high doses. Although probabilities are not considered in these analyses, it should be noted that the total volume of RH waste is only 31 m³, or 0.13% of the volume of the total combined inventory. The probability of a direct interception of RH waste by an intruder is extremely small.

The excavation scenario for the RH waste becomes rather incredible, since excavation of 100 m³ of waste and backfill is assumed, while the total RH waste volume is only 31 m³.

The doses from the intruder-excavation and intruder-agricultural scenarios at 500 yr are dominated by Pu-239 and Am-241. At 500 yr, the Cs-137 (half-life of about 30 yr) has mostly decayed. The doses for these two 500-yr scenarios are similar for the CH, RH, and combined inventory fractions because the concentrations of Pu-239 and Am-241 are identical in the different waste fractions.

The intrusion doses from IEB are approximately 2.3 times higher than the doses from PCC because of the higher density and higher waste loading in IEB.

An additional run was made with IEB for the intruder-agricultural scenario at 100 yr with the 20-ft cover. The calculated dose for the combined RH and CH waste, using the refined inventory, was 97 mrem/yr, just within the performance objective. This result is important because, with the simplified inventory and for IEB at 100 yr with the 20-ft soil cover, the dose was 110 mrem/yr, just above the performance objective. Thus, the refined inventory moves the dose from just above the limit to just below the limit. A similar run for IEB4 at 100 yr with the 20-ft cover gave a dose of 110 mrem/yr.

Because the intruder doses from PCC are lower than those from IEB, the dose from PCC for the intruder-agricultural scenario at 100 yr with the 20-ft cover would be less than 97 mrem/yr. The result would satisfy the performance objective.

Table 10-34. Intruder scenario doses for refined inventory.

FWF	Waste fraction	Intruder-drilling 16-ft soil cover 100 yr (mrem)	Intruder- excavation 16-ft soil cover 500 yr (mrem)	Intruder- agricultural 16-ft soil cover 500 yr (mrem/yr)	Intruder- agricultural 20-ft soil cover 100 yr (mrem/yr)
PCC	CH	2.7	96	1,300	NE ^a
PCC	RH	13,000	92	1,300	NE
PCC	Combined ^b	19	96	1,300	NE
IEB	CH	6.2	220 ^c	3,000	NE
IEB	RH	29,000	210 ^c	3,100	NE
IEB	Combined ^b	43	220 ^c	3,000	97 ^d

a. NE = Not evaluated.

b. The CH waste and RH waste totals for the various nonactinides differed slightly from those in the combined simplified inventory. Therefore, in some instances, the dose for the combined waste with the refined inventory differs slightly from that for the simplified inventory.

c. The small percentage differences in this column are due to rounding of radionuclide inventory values in the calculations.

d. With an IEB4 FWF, the result was 110 mrem/yr.

10.4.7 Effects of Inventory with Additional Refinements on Results

Section 2.4.4 describes the inventory with additional refinements. The inventory contains revised activities of the nonactinides, plus new entries for H-3, C-14, Tc-99, and I-129. All of the H-3 is taken to be in the CH fraction. The C-14, Tc-99, and I-129 are assumed conservatively to have the same activities in the CH, RH, and combined inventories. Compared with the refined inventory discussed in Section 2.4.3, the inventory with additional refinements contains less nonactinide activity in the RH portion, slightly more in the CH portion, and less overall.

Doses were calculated for the intruder-drilling, intruder-excavation, and intruder-agricultural scenarios. The FWFs evaluated were PCC, IEB, and IEB4. The results are presented in Table 10-35. Separate results are shown for the CH, RH, and combined inventory fractions.

For comparison purposes, Table 10-35 also presents the results of the corresponding runs with the simplified inventory and the refined inventory.

Table 10-35. Intruder scenario doses for inventory with additional refinements.

FWF	Inventory ^a	Waste fraction	Intruder-drilling scenario 16-ft soil cover 100 yr (mrem)	Intruder-drilling scenario 20-ft soil cover 100 yr (mrem)	Intruder-excavation scenario 16-ft soil cover 500 yr (mrem)	Intruder-agricultural scenario 16-ft soil cover 500 yr (mrem/yr) ^b	Intruder-agricultural scenario 20-ft soil cover 100 yr (mrem/yr) ^b	Intruder-agricultural scenario 20-ft soil cover 500 yr (mrem/yr) ^b
PCC	IAR	CH	3	2.5	96	1,300	29	9.9
PCC	IAR	RH	2,000	1,700	96	1,400	1,700	10
PCC	IAR	Combined	5.4	3.6	96	1,300	31	9.9
PCC	S	NA ^c	23	18	96	1,400	46	NE ^d
PCC	R	CH	2.7	NE	96	1,300	NE	NE
PCC	R	RH	13,000	NE	92	1,300	NE	NE
PCC	R	Combined	19	NE	96	1,300	NE	NE
IEB	IAR	CH	6.7	5.7	220	3,000	67	22
IEB	IAR	RH	4,500	3,600	220	3,100	3,800	23
IEB	IAR	Combined	15	12	220	3,100	75	22
IEB	S	NA	49	42	220	3,100	110	22
IEB	R	CH	6.2	NE	220	3,000	NE	NE
IEB	R	RH	29,000	NE	210	3,100	NE	NE
IEB	R	Combined	43	NE	220	3,000	97	22
IEB4	IAR	CH	8.4	6.6	260	3,600	78	26
IEB4	IAR	RH	5,500	4,500	260	3,600	4,400	26
IEB4	IAR	Combined	21	17	260	3,600	83	26
IEB4	S	NA	59	49	260	3,600	120	25
IEB4	R	Combined	NE	NE	NE	NE	110	25

a. S = Simplified; R = Refined; IAR = Inventory with additional refinements.

b. The small percentage differences reflected in this column are due to rounding of radionuclide inventory values in the calculations.

c. NA = Not applicable.

d. NE = Not evaluated.

The key conclusions and comparisons from the results in Table 10-35 are as follows:

- With either the refined inventory or the inventory with additional refinements, the RH portion of the waste does not comply with the performance objectives. However, the intrusion scenarios involving the RH waste again become rather incredible because of the small volume of this fraction of the ALLW.
- With the 20-ft soil cover and the inventory with additional refinements, the CH portion and combined portions for all FWFs now comply with all intrusion performance objectives.
- For the inventory with additional refinements, the behavior in terms of dominant radionuclide contributors to dose is generally the same as that for the refined inventory, as discussed in Section 10.4.6.
- H-3 is not an important contributor to the intruder dose because of its relatively short (12.3-yr) half-life. C-14, Tc-99, and I-129 are not important contributors because of their extremely low activities in the INEL ALLW.
- The intruder doses from IEB are approximately 2.3 times higher than those from PCC, and the doses from IEB4 are still higher. This behavior is a consequence of the higher densities and higher waste loadings in these FWFs.

10.4.8 Effects of Exposure Duration on Results for Intruder-Drilling Scenario

The evaluations of the intruder-drilling scenario were based on the assumption that 1 hr would be required to drill through the waste and 40 hr to complete installation of a residential well. The drillers would be exposed to 1 hr of inhalation during the penetration of the waste layer and 40 hr of direct radiation from the drill cuttings. These assumptions are less conservative than the corresponding assumptions in Maheras et al. 1994. That assessment assumed 160 hr of inhalation and 160 hr of external exposure, based on construction of a large irrigation well. See Table H-1 in Appendix H.

A run was performed with an IEB FWF to investigate the effects of more conservative values for the exposure times. The inhalation time was increased from 1 hr to 8 hr. The exposure time for external radiation was left at 40 hr to reflect a residential well.

The results of the run are compared with those from an IEB run with the shorter exposure time in Table 10-36. Both evaluations assume intrusion at 100 yr postclosure, an IEB FWF, and a 20-ft soil cover.

The dose increased by only 2 mrem/yr, or about 5%, when the inhalation exposure was lengthened. Most of the dose results from direct exposure from Cs-137, rather than inhalation of radionuclides (Am-241 being the primary contributor to inhalation dose). Therefore, the assumed duration of the inhalation has only a minor effect on the total dose from all exposure routes in the intruder-drilling scenario.

Table 10-36. Effects of exposure duration in the intruder-drilling scenario. (Both runs are based on an IEB FWF, 20-ft soil cover, and a time of 100 yr postclosure).

Exposure duration	Total dose (mrem)	Radionuclides contributing ≥5% of dose	Radionuclide contribution (%)
1 hr inhalation	42	Cs-137	93
40 hr external		Am-241	7
8 hr inhalation	44	Cs-137	89
40 hr external		Am-241	9

10.4.9 Effects of Post-Disruption Particle Size Distribution of Final Waste Form on Results

10.4.9.1 Introduction. The assumptions and parameter values used for the intrusion scenarios are discussed in Section 9. The size distribution of the particles produced by drilling through or excavation into the disposed waste is an important parameter in modeling those scenarios. Either drilling or excavation could disrupt a portion of the waste, grind it or shear it into smaller fragments, and bring a portion of the broken-up material to the surface.

The presence of the broken-up waste material at the surface is assumed to lead to acute exposure of the drilling and excavation crews by direct radiation and by inhalation. In addition, in the intruder-agricultural scenario, the exhumed material from drilling or excavation is spread over an area where crops are grown. This action results in chronic exposure by inhalation and direct exposure. It also results in chronic exposure by the pathway involving leaching of radionuclides from the FWF into the root zone, uptake of the radionuclides by crops, and human ingestion of the crops or of livestock that feed upon the crops.

10.4.9.2 General Effects of Particle Size. The following paragraphs discuss the general effects of FWF particle size in these three intrusion exposure routes—direct radiation, inhalation, and ingestion of foodstuffs.

For direct radiation exposure, the particle size distribution could have some effect on the dose by virtue of the differing degrees of self-shielding provided by the waste fragments. Large fragments would offer more self-shielding than would fine particulate. However, this effect is expected to be relatively minor, particularly since the fractured FWF would be mixed with surface soil to a depth of 15 cm. This effect was not addressed further.

For inhalation, the particle size distribution could have a major effect on the dose. If the mechanical disruption due to the intrusion produces a large fraction of minute particulate, then a relatively high concentration of airborne, respirable particulate could be present in the air inhaled by the intruders. On the other hand, if the broken-up FWF exists primarily in larger chunks, very little might become airborne and respirable. A simplified evaluation of the effect was undertaken and is described in Section 10.4.9.4.

For the ingestion exposure route, the particle size distribution could have a major effect on root uptake. If the FWF exists in large fragments, root uptake of the radionuclides in the FWF would be much less than if the FWF exists totally as fine particulate. The difference in uptake would be expected to be related to the comparative total surface area of the FWF. This effect would be quite difficult to evaluate by modeling, however. A simplified evaluation of the effect was undertaken and is described in Section 10.4.9.3.

10.4.9.3 Simplified Evaluation of Effects of Particle Size on Ingestion Dose. In the current methodology for intrusion evaluation, the radionuclide concentration in crops is calculated using a "soil-to-plant transfer factor" for each element and each food group. These factors relate the concentrations of elements in soil to the concentrations in agricultural products grown in that soil. For each element, there is a set of factors for leafy vegetables/fresh animal forage, root vegetables, grains/stored animal feed, and fruit. The factors are unitless parameters that can be thought of as (pCi/g in the plant/pCi/g in the soil). These factors have values that range from 10^{-4} to 10^{-1} for radionuclides of concern.

In the current methodology, the values of the factors are based on the radionuclides being freely available in the soil, e.g., as a powder mixed with the soil or even as a dissolved species. These conditions correspond to the maximum potential for uptake by plant roots.

More realistically, the FWF fragments in the intrusion scenario would range in size from large chunks to a relatively small fraction of fine particulate. Postulating plant root uptake from such an assumed distribution of particle sizes would be very complex. Therefore, to investigate the effect of this parameter on the calculated ingestion dose, the simple approach was used of reducing the soil-to-plant uptake factors by a factor of 10 for each FWF. A total of five runs were made with these reduced values. The five runs represent all six FWFs because the characteristics of PCC and FUETAP were assumed to be identical.

10.4.9.4 Simplified Evaluation of Effects of Particle Size on Inhalation Dose.

10.4.9.4.1 Current Calculational Method—Exposure via inhalation of suspended particulates in the intrusion scenarios is calculated using the GENII code. The dose to an individual from inhalation of contaminated air is calculated from the product of the concentration (activity) of each radionuclide in the air, ventilation (breathing) rate, exposure duration, and dose conversion factor. The doses from the various radionuclides are summed. The air activity can be calculated using either a time-dependent function or a constant mass loading factor. The latter choice is used, with a mass loading of 10^{-4} g/m³ for drilling and 5×10^{-3} g/m³ for excavation.

The internal dosimetry model of GENII calculates the dose from inhalation of contamination air using a multicompartmental model. A value of 1.0 micron for the activity median aerodynamic diameter (AMAD) is assumed for all airborne material.

10.4.9.4.2 The Physical Nature of the Actions in the Intrusion Scenarios—In the intruder-drilling scenario, the buried containers of FWF are assumed to be stacked vertically as shown in Figure 5-1. The diameter of each container is approximately 60 cm. A 30-cm-diameter drill bit is assumed to penetrate a column of containers vertically. Depending on the drilling process, the FWF is subjected to a shearing and grinding process, and possibly a pulverizing

process, that generates fragments ranging in size from particulate to large chunks. The total volume of FWF fractured and removed is $(\pi[0.3 \text{ m}]^2/4) (6.9 \text{ m}) = 0.49 \text{ m}^3$. Bringing the fragments to the surface and spreading them out is not expected to alter the size distribution appreciably, but it would dilute the fragments with uncontaminated soil.

In the intruder-excavation scenario, the buried containers of FWF are subjected to a shearing process. The exact nature of the process depends on the excavation method. The extent of FWF breakup into fine particulate is expected to be less than that associated with drilling. For this reason and because penetration of the FWF during excavation can be excluded by disposal at the greater of the two depths studied, the current evaluation concentrated on the drilling action.

10.4.9.4.3 Available Data—No particulate data were identified for conditions that closely match those in the intruder-drilling scenario. Developing rigorous assumptions that cover all of the complexities associated with drilling through a FWF would require an extensive testing program, which is beyond the scope of the present study.

Data from impact testing are available from which a very rough, order-of-magnitude estimate can be developed of the respirable fraction that might result from drilling through a FWF. There is no certainty that the impact data bear a meaningful relationship to the data from drilling through a FWF. Nevertheless, the impact data can be used to illustrate the effects of particle size and airborne concentrations on the calculated doses.

Data are available for the particle size distribution from several FWFs subjected to impact (not shearing and grinding) actions. Smith and Ross (1975) subjected large (5-ft long) and smaller (4-in. long) steel canisters containing a glass FWF to impact at speeds of up to 117 ft/sec. The resulting fraction of the FWF that was of respirable size (by convention, <10 microns) was on the order of 10^{-4} (0.02%) at the highest speed. The energy density at the highest speed was 1.4 J/cm^3 .

Jardine et al. (1982) subjected small (approximately 40-g) specimens of several FWFs to impact from a large falling weight. The energy density for most of the tests was 10 J/cm^3 . The respirable fractions were 0.43% for FUETAP cement, 0.14% to 0.18% for glasses, and 0.06% to 0.16% for ceramics. Limited tests at energy densities as low as 1.2 J/cm^3 suggested that the respirable size fraction increases approximately linearly with increasing density of impact energy. At an energy density of 1.2 J/cm^3 , the respirable fraction from glass specimens was 0.016%. Allowing for the uncertainties of about a factor of 2 upward or downward in the comparison, this value of 0.016% is close to the slightly higher value of 0.02% obtained by Smith and Ross (1975) at a slightly higher energy density.

Farnsworth et al. (1994) reported respirable fractions of 0.32% for PCC and 0 for IEB and IEB4, using the method of Jardine et al. (1982) with 40-g specimens. The energy density was the same 10 J/cm^3 .

A compendium of data on accident-generated particulate materials (Sutter 1982) contains data from repeated impact of weights on small samples of concrete and glass. The energy density

of the cumulative impacts was approximately 25 to 35 J/cm³. No glass particles <8 microns were produced; for concrete, 0.1% to 0.2% of the mass was in particulate <8 microns.

10.4.9.4.4 Simplified Method for Evaluating Effects of Particle Size—Accurate modeling of the distribution of particle sizes inhaled in the intrusion scenarios would be extremely complex. The first step would involve estimating the size distribution of particulate matter produced during drilling, which depends on the FWF, the drill bit material and design, and the drilling procedure. The next step would involve modeling the process by which the particulates become airborne, which depends on the drilling procedure, the provisions for managing the drilling effluents and residues, the wind conditions, the nature of the ground surface, and many other parameters. The deposition of suspended particles, at a rate that depends on particle size, would also be modeled. The locations and work habits of the drilling personnel relative to the drilling apparatus would also be modeled. The result of this complex set of analyses would be a time-dependent, particle size-dependent air loading at the locations where the airborne particles could be inhaled by the drilling personnel. Unfortunately, only very simplified, approximate models of the above conditions are feasible. One such model is as follows.

Based on the impact tests, an order-of-magnitude estimate for the respirable fractions (<10 microns) of FWF that might result from drilling through full-size containers of waste is 0.5% for the cements, 0.2% for the glasses, and 0.1% for ceramics. Based on the particle size distribution data of Smith and Ross (1975), the <5 micron fraction would be about a factor of 10 smaller than the <10 micron fraction, and the <1 micron fraction would be approximately a factor of 100 still smaller.

As shown in Smith and Ross (1975), most particles larger than 7 microns AMAD tend to be trapped in the nasopharyngeal region rather than in the lung (where the resulting dose would be maximized). As the particle size decreases, the portion of the particles trapped in the tracheobronchial and pulmonary (deep lung) compartments increases. For a simple scoping calculation, a fraction of 0.01% of the FWF, corresponding to the <5 micron fraction of ceramics, was used for all FWFs.

Sutter (1982) provides a comprehensive compilation of data on the resuspension factor, k , for particulate matter under a wide range of conditions:

$$k \text{ m}^{-1} = \text{airborne concentration/m}^3 / \text{surface contamination/m}^2.$$

For outdoor, cleared sandy soil at wind speeds <5 m/sec, k typically ranges from 10^{-10} to 10^{-6} m^{-1} .

In GENII, the surface areas over which the excavated material is spread are assumed to be 5.5 m^2 for the drilling scenario and $1,680 \text{ m}^2$ for the excavation scenario.

The airborne concentration of <5 micron particles could be estimated in each scenario. For drilling, assuming an exhumed volume of 0.49 m^3 , the mass of the <5 micron fraction for IEB would be:

$$0.49 \text{ m}^3 \times 10^6 \text{ cm}^3/\text{m}^3 \times 3 \text{ g/cm}^3 \text{ (IEB density)} \times 10^{-4} \text{ (fraction respirable and trapped in lung)} = 147 \text{ g}$$

If this mass were spread over 5.5 m^2 (the area for the drilling scenario), the concentration would be 26.7 g/m^2 . Based on a resuspension factor of 10^{-7} m^{-1} , the mass loading factor would be $2.7 \times 10^{-6} \text{ g/m}^3$, compared with 10^{-4} g/m^3 used previously. Similar calculations were done for the other FWFs by changing the density. For IEB4, BSG, ASG, and FUETAP (or PCC), the mass loading factors are 3.2×10^{-6} , 2.4×10^{-6} , 2.4×10^{-6} , and $1.8 \times 10^{-6} \text{ g/m}^3$, respectively. These mass loading factors were applied to both the drilling and agricultural scenarios.

10.4.9.5 Results of Runs to Evaluate Effects of Particle Size Distribution. Based on the revised values for the mass loading (from Section 10.4.9.4.4) and the plant-soil uptake factors (one tenth of the previous values), new runs were performed for IEB, ASG, IEB4, BSG, and FUETAP (or PCCF) FWFs, for the intruder-drilling and intruder-agricultural scenarios.

Table 10-37 presents the original and revised doses for the intruder-drilling scenario at 100 yr with a 20-ft cover. Table 10-38 presents the original and revised doses for the intruder-agricultural scenario for the same time and cover thickness. The doses for the revised runs were calculated using the revised mass-loading and plant-soil uptake factors.

Table 10-37. Effects of particle size distribution on results for intruder-drilling scenario for 20-ft cover, 100 yr.

Final waste form	Run	Mass loading factor (g/m^3)	Inhalation dose (mrem)	Ingestion dose (mrem)	External dose (mrem)	Total dose (mrem) ^a	Radionuclides contributing $\geq 5\%$ of dose	Radionuclide contribution ^a (%)
IEB	Original	10^{-4}	0.25	0	42	42	Cs-137 Am-241	92 7
	Revised	2.7×10^{-6}	6.9×10^{-3}	0	42	42	Cs-137 Am-241	92 7
ASG	Original	10^{-4}	0.23	0	36	36	Cs-137 Am-241	92 7
	Revised	2.4×10^{-6}	5.4×10^{-3}	0	36	36	Cs-137 Am-241	92 7
IEB4	Original	10^{-4}	0.3	0	49	49	Cs-137 Am-241	90 9
	Revised	3.2×10^{-6}	9.7×10^{-3}	0	49	49	Cs-137 Am-241	90 9
BSG	Original	10^{-4}	0.11	0	18	18	Cs-137 Am-241	93 7
	Revised	2.4×10^{-6}	2.7×10^{-3}	0	18	18	Cs-137 Am-241	93 7
FUETAP or PCC	Original	10^{-4}	0.11	0	18	18	Cs-137 Am-241	93 7
	Revised	1.8×10^{-6}	2.0×10^{-3}	0	18	18	Cs-137 Am-241	93 7

a. The sum of the individual doses may not equal the total dose due to rounding.

Table 10-38. Effects of particle size distribution on results for intruder-agricultural scenario for 20-ft cover, 100 yr.

Final waste form	Run	Mass loading factor (g/m ³)	Inhalation dose (mrem)	Ingestion dose (mrem)	External dose (mrem)	Total dose (mrem) ^a	Radionuclides contributing ≥5% of dose	Radionuclide contribution ^a (%)
IEB	Original	10 ⁻⁴	57	23	26	110	Am-241	40
							Cs-137	22
							Sr-90	14
							Pu-239	11
							Pu-238	7
	Revised	2.7 × 10 ⁻⁶	1.5	2.6	26	30	Cs-137	85
ASG	Original	10 ⁻⁴	50	19	23	92	Am-241	10
							Cs-137	5
							Sr-90	40
							Pu-239	23
							Pu-238	14
	Revised	2.4 × 10 ⁻⁶	1.2	2.2	23	26	Cs-137	12
IEB4	Original	10 ⁻⁴	66	25	32	120	Am-241	7
							Cs-137	84
							Sr-90	10
							Pu-239	5
							Pu-238	40
	Revised	3.2 × 10 ⁻⁶	2.1	2.9	32	37	Cs-137	25
BSG	Original	10 ⁻⁴	25	9.9	12	46	Am-241	14
							Cs-137	39
							Sr-90	23
							Pu-239	14
							Pu-238	11
	Revised	2.4 × 10 ⁻⁶	0.59	1.1	12	14	Cs-137	7
FUETAP or PCC	Original	10 ⁻⁴	25	9.4	12	46	Am-241	84
							Cs-137	10
							Sr-90	5
							Pu-239	39
							Pu-238	24
	Revised	1.8 × 10 ⁻⁶	0.44	1.0	12	13	Cs-137	13

a. The sum of the individual doses may not equal the total dose due to rounding.

For the intruder-drilling scenario (see Table 10-37), the decrease in the mass loading factor lowered the inhalation dose by a factor of about 40. However, the total dose was dominated by the external dose, and the change in the inhalation dose did not have a substantial effect on the total dose. Hence, for this scenario, the total dose for each FWF was unchanged by the refined assumptions regarding size distribution.

For the intruder-agricultural scenario (see Table 10-38), the decreases in the plant-soil uptake factor and the mass loading factor had a substantial effect on the total doses for all the FWFs. The inhalation doses and ingestion doses decreased by approximately the same fraction as the reduction in the mass loading factors and the plant-soil uptake factor, respectively. These values were a factor of about 10 reduction in the ingestion dose and a factor of about 40 reduction in the inhalation dose. The external doses remained the same. Because the inhalation and ingestion doses were major contributors for this scenario, the total dose changed considerably.

The most important result of the refined treatment of particle size distribution is that the doses for all of the FWFs now satisfy the performance objectives. In particular, the intruder-agricultural scenario doses for IEB and IEB4 were reduced from values >100 mrem/yr to values substantially <100 mrem/yr.

10.4.10 Effects of Scenario Timing on Dose

10.4.10.1 Introduction. For all the previous intrusion scenarios, the time of intrusion was assumed to be either 100 or 500 yr after disposal of the waste and closure of the disposal facility. As explained in Section 10.4.1, the 100-yr time was chosen because it corresponds to the start of the postinstitutional control period. The 500-yr time represents the lifetime of the abovegrade cover over the waste. Additional intrusion simulations were performed at other times to confirm the validity of the assumption that the 100-yr and 500-yr times encompass the peak dose for all of the scenarios studied. The results are reported herein.

To determine when the peak intrusion doses occur, five sets of runs were performed. All five sets used the IEB4 FWF, since IEB4 is the most concentrated FWF of those considered and gives the highest dose.

For the first four sets of runs, the burial depth was 5 ft below grade. This results in a total of 16 ft of material over the waste for the first 500 yr (5 ft below grade and 11 ft abovegrade). After 500 yr, only the 5 ft of belowgrade material remains. For the fifth set of runs, the burial depth was 9 ft, resulting in a total thickness of 20 ft for the first 500 yr and 9 ft thereafter.

GENII was used to model the three exposure pathways: external radiation, inhalation, and ingestion. However, at times of intrusion greater than 100,000 yr, the GENII computer model does not calculate inhalation or ingestion pathways; therefore, its results are not valid beyond this point.

For each radionuclide, GENII contains three separate dose conversion factors (one for each pathway). At intrusion times greater than 5,000 yr, doses for the INEL ALLW are dominated by Th-229 or Ac-225. Th-229 has higher dose conversion factors than Ac-225 for the inhalation and ingestion pathways, but Ac-225 has a higher dose conversion factor than Th-229 for the external

pathway. The intruder-drilling scenario is dominated by the external pathway, so, at times of intrusion greater than 5,000 yr, Ac-225 dominates the dose. The intruder-excavation and intruder-agricultural scenarios are dominated by the inhalation pathway, so, at times of intrusion greater than 5,000 yr, Th-229 dominates the dose.

10.4.10.2 Intruder-Drilling Scenario. The first set of runs is the intruder-drilling scenario. The original run for this set had a time of intrusion at 100 yr. Table 10-39 contains the results of nine runs at other intrusion times as well as the original run.

The maximum dose is from the original run, confirming the assumptions made in Section 10.4.1.2. The dose decreases rapidly as the time of intrusion increases. The dominant contributor in the original and 200-yr runs is Cs-137. Cs-137 has a half-life of 30 yr, so it decays relatively fast. By the next run at 300 yr, the Cs-137 has decayed sufficiently that it is no longer the dominant contributor. Am-241 is the dominant contributor for the intrusion times from 300 through 1,000 yr. After 1,000 yr, the Am-241 has decayed sufficiently that one of its progeny, Ac-225, dominates the dose for the rest of the runs. For the intruder-drilling scenario, the dose from external exposure is greater than the dose from inhalation.

Similar intruder-drilling runs over a range of times were not made for the case of the 20-ft cover. The doses would be smaller than those with the 16-ft cover, all of which complied with the performance objective.

10.4.10.3 Intruder-Excavation Scenario. The second set of runs is the intruder-excavation scenario. The original run for this set had a time of intrusion at 500 yr. Table 10-40 contains the results of nine runs at other intrusion times as well as the original run. No runs were

Table 10-39. Time dependence of dose in intruder-drilling scenario with 16-ft soil cover.

Time of intrusion (yr)	Total dose (mrem)	Dominant contributor to dose
100 (original run)	59	Cs-137
200	8.4	Cs-137
300	3	Am-241
400	2.1	Am-241
500	1.6	Am-241
1,000	0.49	Am-241
5,000	0.021	Ac-225 (Am-241 decay chain)
10,000	9.6E-4	Ac-225 (Am-241 decay chain)
50,000	1.3E-14	Ac-225 (Am-241 decay chain)
100,000	3.6E-28	Ac-225 (Am-241 decay chain)

Table 10-40. Time dependence of dose in intruder-excavation scenario with 16-ft soil cover.

Time of simulation (yr)	Total dose (mrem)	Dominant contributor to dose
500 (original run)	260	Pu-239
600	210	Pu-239
700	180	Pu-239
800	150	Pu-239
900	120	Pu-239
1,000	100	Pu-239
5,000	0.61	Pu-239
10,000	1.1E-2	Th-229 (Am-241 decay chain)
50,000	1.5E-13	Th-229 (Am-241 decay chain)
100,000	4.1E-27	Th-229 (Am-241 decay chain)

made for times <500 yr because the abovegrade cover would still be intact and the excavation would not penetrate the waste.

The maximum dose is from the original run, confirming the assumptions made in Section 10.4.1.3. The dose decreases as the time of intrusion increases. The dominant contributor in the original case is Pu-239. Pu-239 dominates the dose in the simulations up to 10,000 yr. Beginning at the 10,000-yr time of intrusion, Th-229 dominates the dose. Th-229 is in the decay chain of Am-241 and has a half-life of 7,430 yr. For the intruder-excavation scenario, the dose from inhalation is the dominant exposure pathway.

Similar intruder-excavation runs were not made for the case of the 20-ft cover because, even after loss of the abovegrade portion of the cover at 500 yr, the excavation would not penetrate the waste.

10.4.10.4 Intruder-Agricultural Scenario. The third set of runs is the intruder-agricultural scenario initiated by drilling with a 16-ft cover. The original run for this set had a time of intrusion at 100 yr. Table 10-41 contains the results of three runs at later times as well as the original run.

The maximum dose is from the original run, confirming the assumptions made in Section 10.4.1.4. The dose decreases as the time of intrusion increases. The dose from the agricultural scenario initiated by drilling is always less than the dose from the agricultural scenario initiated by excavation (assuming the excavation penetrates the waste) because of the small volume of waste exhumed by the drill bit. The dominant contributor in these simulations is Am-241. For the intruder-agricultural scenario, inhalation is the dominant exposure pathway.

Table 10-41. Time dependence of dose in intruder-agricultural scenario initiated by drilling with 16-ft soil cover.

Time of simulation (yr)	Total dose (mrem/yr)	Dominant contributor to dose
100 (original run)	120	Am-241
200	59	Am-241
300	43	Am-241
400	33	Am-241

The fourth set of runs is the intruder-agricultural scenario initiated by excavation with a 16-ft cover. The original run for this set had a time of intrusion at 500 yr. Table 10-42 contains the results of five runs at longer times as well as the original run.

The maximum dose is from the original run, confirming the assumptions made in Section 10.4.1.4. The dose decreases rapidly as the time of intrusion increases. The dominant dose contributor of the original dose is from Am-241. For the 1,000- and 5,000-yr runs, Pu-239 dominates the dose. For the runs from 10,000 to 100,000 yr, Th-229 dominates the dose. Th-229 is produced by the Am-241 decay chain. For the intruder-agricultural scenario, inhalation is the dominant exposure pathway.

The fifth set of runs is the intruder-agricultural scenario initiated by drilling with a 20-ft cover. The original run for this set had a time of intrusion at 500 yr. Table 10-43 contains the results of nine runs at longer times as well as the original run.

The maximum dose is from the original run, confirming the assumptions made in Section 10.4.1.4. The dose decreases rapidly as the time of intrusion increases. The dominant dose contributor up through 500 yr is Am-241. For the 1,000 and 5,000-yr runs, Pu-239 dominates the dose. For the runs from 10,000 to 100,000 yr, Th-229 dominates the dose. Inhalation, initiated by drilling, is the dominant exposure pathway.

Thus, all of these results confirmed the earlier reasoning in Section 10.4.1 that concluded the highest dose to intruders occurs at the 100-yr loss of institutional control or the 500-yr loss of the abovegrade portion of the cover, depending on the specific scenario.

10.4.11 Dominant Radionuclide Contributors to Dose

The dominant radionuclides contributing to dose for the three intrusion scenarios that have been the principal focus of Section 10.4 are shown in Table 10-44. Contributions rounded to the nearest whole percent are provided for each radionuclide that contributes 5% or more to the total dose.

To keep Table 10-44 to a manageable size, the detailed breakdowns of the dominant radionuclide contributors to dose are not listed for certain runs. This restriction applies to the runs made to determine the time to peak dose in Section 10.4.10; the one dominant contributor to each of these runs is given already in Tables 10-39 through 10-43. It also applies to the runs in

Table 10-42. Time dependence of dose in intruder-agricultural scenario initiated by excavation with 16-ft soil cover.

Time of intrusion (yr)	Total dose (mrem/yr)	Dominant contributor to dose
500 (original run)	3,600	Am-241
1,000	1,300	Pu-239
5,000	6.9	Pu-239
10,000	1.6E-1	Th-229 (Am-241 decay chain)
50,000	2.2E-12	Th-229 (Am-241 decay chain)
100,000	6.1E-26	Th-229 (Am-241 decay chain)

Table 10-43. Time dependence of dose in intruder-agricultural scenario initiated by drilling with 20-ft soil cover.

Time of simulation (yr)	Total dose (mrem/yr)	Dominant contributor to dose
100 (original run)	120	Am-241
200	59	Am-241
300	43	Am-241
400	33	Am-241
500	25	Am-241
1,000	9.1	Pu-239
5,000	4.9E-2	Pu-239
10,000	1.1E-3	Th-229 (Am-241 decay chain)
50,000	1.6E-14	Th-229 (Am-241 decay chain)
100,000	4.4E-28	Th-229 (Am-241 decay chain)

Table 10-44. Dominant radionuclides contributing to dose for the intrusion scenarios.

Final waste form	Scenario	Inventory ^a	Total dose (mrem) (mrem/yr for agricultural)	Radionuclides contributing $\geq 5\%$ of dose ^b	Radionuclide contribution ^c (%)
PCC or FUETAP	Intruder-drilling 16-ft cover after 100 yr	Simplified	23	Cs-137	93
				Am-241	7
PCC or FUETAP	Intruder-drilling 20-ft cover after 100 yr	Simplified	18	Cs-137	93
				Am-241	7
PCC or FUETAP	Intruder-excavation 16-ft cover after 500 yr	Simplified	96	Pu-239	49
				Am-241	36
				Pu-240	11
PCC or FUETAP	Intruder-agricultural 16-ft cover after 500 yr	Simplified	1,400	Am-241	57
				Pu-239	31
				Pu-240	7
PCC or FUETAP	Intruder-agricultural 20-ft cover after 100 yr	Simplified	46	Am-241	39
				Cs-137	24
				Sr-90	13
				Pu-239	11
				Pu-238	7
IEB	Intruder-drilling 16-ft cover after 100 yr	Simplified	49	Cs-137	90
				Am-241	7
IEB	Intruder-drilling 20-ft cover after 100 yr	Simplified	42	Cs-137	93
				Am-241	7
IEB	Intruder-excavation 16-ft cover after 500 yr	Simplified	220	Pu-239	50
				Am-241	36
				Pu-240	11
IEB	Intruder-agricultural 16-ft cover after 500 yr	Simplified	3,100	Am-241	58
				Pu-239	30
				Pu-240	7
IEB	Intruder-agricultural 20-ft cover after 100 yr	Simplified	110	Am-241	40
				Cs-137	22
				Sr-90	14
				Pu-239	10
				Pu-238	7
IEB	Intruder-agricultural 20-ft cover after 500 yr	Simplified	22	Am-241	59
				Pu-239	30
				Pu-240	7
IEB	Intruder-drilling 16-ft cover after 100 yr	Simplified with 10 nCi/g TRU	45	Cs-137	98
IEB	Intruder-drilling 20-ft cover after 100 yr	Simplified with 10 nCi/g TRU	39	Cs-137	100
IEB	Intruder-excavation 16-ft cover after 500 yr	Simplified with 10 nCi/g TRU	22	Pu-239	50
				Am-241	36
				Pu-240	11

Table 10-44. (continued).

Final waste form	Scenario	Inventory ^a	Total dose (mrem) (mrem/yr for agricultural)	Radionuclides contributing $\geq 5\%$ of dose ^b	Radionuclide contribution ^c (%)
IEB	Intruder-agricultural 16-ft cover after 500 yr	Simplified with 10 nCi/g TRU	310	Am-241	58
				Pu-239	31
				Pu-240	7
IEB	Intruder-agricultural 20-ft cover after 100 yr	Simplified with 10 nCi/g TRU	47	Cs-137	51
				Sr-90	32
				Am-241	9
IEB	Intruder-drilling 20-ft cover after 100 yr	Simplified with 1000 nCi/g TRU	84	Cs-137	52
				Am-241	44
IEB	Intruder-drilling 20-ft cover after 100 yr	Simplified with 1000 nCi/g TRU	72	Cs-137	54
				Am-241	43
IEB	Intruder-excavation 16-ft cover after 500 yr	Simplified with 1000 nCi/g TRU	2,200	Pu-239	50
				Am-241	36
				Pu-240	11
IEB	Intruder-agricultural 16-ft cover after 500 yr	Simplified with 1000 nCi/g TRU	31,000	Am-241	58
				Pu-239	31
				Pu-240	7
IEB	Intruder-agricultural 20-ft cover after 100 yr	Simplified with 1000 nCi/g TRU	690	Am-241	62
				Pu-239	16
				Pu-238	11
IEB4	Intruder-drilling 16-ft cover after 100 yr	Simplified	59	Cs-137	92
				Am-241	7
IEB4	Intruder-drilling 20-ft cover after 100 yr	Simplified	49	Cs-137	90
				Am-241	8
IEB4	Intruder-excavation 16-ft cover after 500 yr	Simplified	260	Pu-239	46
				Am-241	37
				Pu-240	11
IEB4	Intruder-agricultural 16-ft cover at 500 yr	Simplified	3,600	Am-241	70
				Pu-239	30
IEB4	Intruder-agricultural 20-ft cover after 100 yr	Simplified	120	Am-241	40
				Cs-137	25
				Sr-90	14
				Pu-239	11
				Pu-238	7
IEB4	Intruder-agricultural 20-ft cover after 500 yr	Simplified	25	Am-241	60
				Pu-239	32
				Pu-240	8
BSG	Intruder-drilling 16-ft cover at 100 yr	Simplified	23	Cs-137	91
				Am-241	7

Table 10-44. (continued).

Final waste form	Scenario	Inventory ^a	Total dose (mrem) (mrem/yr for agricultural)	Radionuclides contributing $\geq 5\%$ of dose ^b	Radionuclide contribution ^c (%)
BSG	Intruder-drilling 20-ft cover after 100 yr	Simplified	18	Cs-137 Am-241	93 7
BSG	Intruder-excavation 16-ft cover after 500 yr	Simplified	96	Pu-239 Am-241 Pu-240	49 36 11
BSG	Intruder-agricultural 16-ft cover at 500 yr	Simplified	1,400	Am-241 Pu-239 Pu-240	57 31 7
BSG	Intruder-agricultural 20-ft cover after 100 yr	Simplified	46	Am-241 Cs-137 Sr-90 Pu-239 Pu-238	39 23 14 11 7
ASG	Intruder-drilling 16-ft cover after 100 yr	Simplified	46	Cs-137 Am-241	91 7
ASG	Intruder-drilling 20-ft cover after 100 yr	Simplified	36	Cs-137 Am-241	92 7
ASG	Intruder-excavation 16-ft cover after 500 yr	Simplified	190	Pu-239 Am-241 Pu-240	50 37 11
ASG	Intruder-agricultural 16-ft cover after 500 yr	Simplified	2,700	Am-241 Pu-239 Pu-240	57 31 7
ASG	Intruder-agricultural 20-ft cover after 100 yr	Simplified	92	Am-241 Cs-137 Sr-90 Pu-239 Pu-238	40 23 14 12 7
Untreated	Intruder-drilling 16-ft cover after 100 yr	Refined	12	Cs-137 Am-241	92 8
Untreated	Intruder-drilling 20-ft cover after 100 yr	Refined	9.6	Cs-137 Am-241	90 9
Untreated	Intruder-excavation 16-ft cover after 500 yr	Refined	58	Pu-239 Am-241 Pu-240	50 36 12
Untreated	Intruder-agricultural 16-ft cover after 500 yr	Refined	820	Am-241 Pu-239 Pu-240	60 32 7

Table 10-44. (continued).

Final waste form	Scenario	Inventory ^a	Total dose (mrem) (mrem/yr for agricultural)	Radionuclides contributing $\geq 5\%$ of dose ^b	Radionuclide contribution ^c (%)
Untreated	Intruder-agricultural 20-ft cover after 100 yr	Refined	24	Am-241	46
				Sr-90	14
				Cs-137	13
				Pu-239	13
				Pu-238	9
PCC	Intruder-drilling 16-ft cover after 100 yr	Refined-RH	13,000	Cs-137	99
PCC	Intruder-excavation 16-ft cover after 500 yr	Refined-RH	92	Pu-239	48
				Am-241	36
				Pu-240	12
PCC	Intruder-agricultural 16-ft cover after 500 yr	Refined-RH	1,300	Am-241	57
				Pu-239	31
				Pu-240	7
PCC	Intruder-drilling 16-ft cover after 100 yr	Refined-CH	2.7	Am-241	59
				Cs-137	37
PCC	Intruder-excavation 16-ft cover after 500 yr	Refined-CH	96	Pu-239	49
				Am-241	36
				Pu-240	11
PCC	Intruder-agricultural 16-ft cover after 500 yr	Refined-CH	1,300	Am-241	61
				Pu-239	33
				Pu-240	6
PCC	Intruder-drilling 16-ft cover after 100 yr	Refined-combined	19	Cs-137	89
				Am-241	8
PCC	Intruder-excavation 16-ft cover after 500 yr	Refined-combined	96	Pu-239	49
				Am-241	36
				Pu-240	11
PCC	Intruder-agricultural 16-ft cover after 500 yr	Refined-combined	1,300	Am-241	61
				Pu-239	33
				Pu-240	6
IEB	Intruder-drilling 16-ft cover after 100 yr	Refined-RH	29,000	Cs-137	100
IEB	Intruder-excavation 16-ft cover after 500 yr	Refined-RH	210	Pu-239	47
				Am-241	36
				Pu-240	11
IEB	Intruder-agricultural RH-waste 16-ft cover after 500 yr	Refined-RH	3,100	Am-241	55
				Pu-239	30
				Pu-240	7

Table 10-44. (continued).

Final waste form	Scenario	Inventory ^a	Total dose (mrem) (mrem/yr for agricultural)	Radionuclides contributing $\geq 5\%$ of dose ^b	Radionuclide contribution ^c (%)
IEB	Intruder-drilling 16-ft cover after 100 yr	Refined-CH	6.2	Am-241	60
				Cs-137	35
IEB	Intruder-excavation 16-ft cover after 500 yr	Refined-CH	220	Pu-239	50
				Am-241	36
				Pu-240	11
IEB	Intruder-agricultural 16-ft cover after 500 yr	Refined-CH	3,000	Am-241	60
				Pu-239	31
				Pu-240	7
IEB	Intruder-drilling 16-ft cover after 100 yr	Refined-combined	43	Cs-137	91
				Am-241	9
IEB	Intruder-excavation 16-ft cover after 500 yr	Refined-combined	220	Pu-239	50
				Am-241	36
				Pu-240	11
IEB	Intruder-agricultural 16-ft cover after 500 yr	Refined-combined	3,000	Am-241	60
				Pu-239	31
				Pu-240	7
IEB	Intruder-agricultural 20-ft cover after 100 yr	Refined-combined	97	Am-241	48
				Cs-137	14
				Sr-90	14
				Pu-239	13
				Pu-238	9
IEB	Intruder-agricultural 20-ft cover after 500 yr	Refined-combined	22	Am-241	59
				Pu-239	30
				Pu-240	7
IEB4	Intruder-agricultural 20-ft cover after 100 yr	Refined-combined	110	Am-241	48
				Sr-90	15
				Cs-137	14
				Pu-239	14
				Pu-238	9
IEB4	Intruder-agricultural 20-ft cover after 500 yr	Refined-combined	25	Am-241	60
				Pu-239	32
				Pu-240	8
IEB	Intruder-drilling 20-ft cover after 100 yr increased exposure time	Simplified	44	Cs-137	89
				Am-241	9

a. Unless otherwise indicated, all inventories had a TRU concentration of 100 nCi/g.

b. Listed contributions from actinides include the ingrowth of progeny.

c. Percent contribution of each radionuclide contributing $\geq 5\%$, rounded to nearest whole percent.

Section 10.4.9 that evaluated the effects of particle size distribution, for the same reason. In addition, the dominant contributors for the runs in Section 10.4.7, using the inventory with additional refinements, are not given in Table 10-44. The key contributors are nearly the same as those for the corresponding runs made with the refined inventory.

The intrusion scenarios can be divided into two general categories with respect to the dominant radionuclides that contribute to dose. The doses resulting from intrusion scenarios assumed to occur at 100 yr were dominated by Cs-137 and the Am-241 decay chain. Minor contributors in some scenarios included Sr-90 and the Pu-238, Pu-239, and Pu-240 decay chains. The doses resulting from intrusion scenarios assumed to occur at 500 yr were dominated by the Am-241 chain and the Pu-239 chain. The Pu-240, Pu-238, and U-233 chains contributed lesser percentages, depending on the specific assumptions.

The separate analyses of RH and CH fractions for the refined inventory included the radionuclides Ce-144, Cs-134, Kr-85, Ni-63, Pm-147, and Pr-144, which were excluded in the other analyses. These additional radionuclides, however, contributed much less than 1% of the total dose. For the inventory with additional refinements, which included even more radionuclides, the conclusions were essentially the same.

10.4.12 Results for Intruder-Radon Scenario

Table 10-45 contains the results for the intruder-radon scenario. It indicates the time of maximum dose, the indoor Rn-222 concentration at the time of maximum dose, and the maximum dose from radon inhalation for the PCC and IEB4 FWFs.

The maximum radon dose for the PCC FWF occurs at 4,600 yr. The maximum indoor radon concentration is 1.03×10^{-3} pCi/L. The IEB4 FWF maximum dose occurs at 20,000 yr, with a maximum indoor radon concentration of 1.7×10^{-2} pCi/L. The maximum doses for PCC and IEB4 are 0.19 and 3.16 mrem/yr, respectively.

The time of maximum dose for IEB4 is much later than for PCC because of its much smaller leach rate. The radionuclides in the IEB4 FWF remain in the disposal facility much longer; therefore, the radionuclides undergo more decay and more radon is produced in the facility. This accounts for the higher dose from IEB4.

In Section 8, the peak radon concentration from PCC was found to occur at 167,000 yr, for an intruder standing at the surface. This peak time assumes no leaching of the radionuclides. Because the intruder-radon scenario includes leaching, no radionuclides remain in the facility at 167,000 yr.

In the case of PCC, it was assumed that all of the radon generated is available for transport via diffusion from the porous waste form toward the receptor. In the case of IEB4, the impermeable nature of the FWF prevents radon produced within the FWF from being released from the FWF before it decays into nongaseous products. Therefore, the source of radon for IEB4 was restricted to the inventory of radionuclides present in a leached form within the disposal unit.

Table 10-45. Results for the intruder-radon scenario.

Waste form	Time of maximum dose (yr)	Indoor Rn-222 concentration at time of maximum dose	
		(pCi/L)	Maximum dose (mrem/yr)
PCC	4,600	1.03×10^{-3}	0.19
IEB4	20,000	1.7×10^{-2}	3.16

The PCC leach rate dictates that all the radionuclides are released from the FWF in about 3,300 yr. The maximum dose occurs after this time due to adsorption of radionuclides within the facility. The RESRAD code used for the intruder-radon scenario includes a desorption model to simulate leaching. Leach rates input to the code are converted to equivalent desorption coefficients that are applied equally to the waste, backfill, and vadose zone materials. Thus, sorption in the facility cannot be avoided, but in radon scenarios, incorporating the sorption produces conservative results.

Table 10-46 compares the calculated radon concentrations and doses against regulatory limits. The conclusion is that no limits would be exceeded.

10.5 Qualitative Prediction of the Effects of Secondary FWFs

The evaluations of the FWFs considered in this report are based on the simplifying assumption that all of the waste will be disposed of as a single FWF (PCC, FUETAP, BSG, ASG, IEB, IEB4, or untreated). The selected FWF will be produced by immobilization of the ash remaining after thermal treatment of the waste.

In reality, not all of the existing INEL ALLW waste can be incorporated into a single FWF. It is likely that multiple FWFs will be disposed of. It is also likely that a small fraction of the waste will not be susceptible to thermal treatment and will be incorporated into some secondary waste form not considered in this report. This section discusses the likely effects of such a secondary waste form on the performance of the disposal system.

The total volume of waste that may be incorporated into a secondary waste form (such as a cement waste form) is believed to be in the range of 1% to 5%—certainly less than 10%—of the total volume. Most of this material contains hazardous metals with low melting points, such as lead or mercury, that are not amenable to thermal treatment.

The TRU radionuclide concentration in INEL ALLW is limited, by definition, to a range from 10 through 100 nCi/g. Because of this narrow range, the TRU activity does not concentrate to a great extent in any content code. Therefore, if 10% of the waste volume is present as a secondary waste form, this volume is expected to contain a similar percentage of the total TRU radionuclide inventory.

Table 10-46. Regulatory requirements and results for intruder-radon scenario.

Regulation	Exposure group	Performance objective limit	Results of evaluations
EPA (1987)	Inadvertent intruder	4 pCi/L concentration in indoor air	Maximum indoor air concentration of 1.7×10^{-2} pCi/L for IEB4
DOE Order 5820.2A, Ch. III	Inadvertent intruder	100 mrem/yr EDE (chronic exposure)	Maximum dose of 3.16 mrem/yr for IEB4

The dose for the undisturbed-waste scenario—groundwater transport pathway depends on the FWF. For Site A, for example, the dose varies from 3.6 mrem/yr for untreated waste (Run 1UTR) to 0.011 mrem/yr for IEB4 (Run 3). These doses are all below the 25-mrem/yr performance objective. If the secondary waste form contains 10% of the radionuclide inventory and performs at least as well as the untreated waste, then the presence of a secondary waste form should not pose any problem in demonstrating compliance with the performance objective.

The acute and chronic intrusion doses depend on the volumetric concentrations of radionuclides in the waste form. The concentration in a secondary waste form cannot realistically fall far outside the range of concentrations already considered in the six FWFs and untreated waste, while still meeting the definition of ALLW. Therefore, an inadvertent intrusion into a secondary waste form is expected to result in doses within the same range as the intrusion doses predicted herein.

In summary, the probable presence of a secondary waste form occupying up to 10% of the total waste volume will not significantly affect the overall performance of the disposal system unless the performance of the secondary waste form is a great deal worse than the performance of untreated waste.

References for Section 10

- Allard, B., 1985, *Radionuclide Sorption on Concrete*, Nationale Genossenschaft für die Lagerung radioaktiver Abfälle. NAGRA-NTB-85-21, November.
- Apel, M. L., G. K. Becker, Z. K. Ragan, J. Frasure, B. D. Raivo, L. G. Gale, and D. P. Pace, 1994, *Radiological, Physical, and Chemical Characterization of Low-level Alpha-contaminated Wastes Stored at the Idaho National Engineering Laboratory for Use in Conducting Feasibility Studies for Treatment Services*, EG&G Idaho, EGG-RWMC-11189, March.
- Brandstetter, E.R., J.L. Lolcama, and S.R. Reed, 1994, "An Integrated Degradation and Structural Model for Predicting the Service Life of Buried Reinforced Concrete Structures for Low- and Intermediate-Level Radioactive Waste Disposal," *Proceedings of the Symposium on Waste Management*, Tucson, AZ, February 27-March 3, 1994, Vol. 1, pp. 89-97.
- DOE, 1989, *Technical Approach Document*, UMTRA-DOE/AL 050425.0002, Rev. II, U.S. Department of Energy.
- EPA, 1987, *Interim Protocols for Screening and Follow-up Radon and Radon Decay Product Measurements*, Office of Radiation Programs, EPA 520/1-86-014-1, U.S. Environmental Protection Agency.
- Farnsworth, R. K., et al., 1994, "Properties Comparison for Mixed Waste Final Waste Forms," presented at the American Chemical Society Meeting, September 19-21.
- Glasser, F. P., M. J. Angus, C. E. McCulloch, D. Macphee, and A. A. Rahman, 1985, "The Chemical Environment in Cements," *Material Research Society Symposium Proceedings*, Vol. 44, 1985, pp. 849-858.
- Glasstone, S., and A. Sesonke, 1967, *Nuclear Reactor Engineering*, Van Nostrand Reinhold Company, New York, NY.
- Jardine, L. J., G. T. Reedy, and W. J. Mecham, 1982, "Respirable Fines Produced by Impacts of Simulated Alternative High-Level Waste Materials," in *The Scientific Basis for Nuclear Waste Management*, S. V. Topp, ed., Elsevier Science Publishing Company, Inc., 1982, pp. 115-123.
- Lodge, J. P., Jr., 1991, *Methods of Air Sampling and Analysis*, 3rd ed., Lewis Publishers, Inc., Chelsea, Michigan.
- Lutze, W., and R. C. Ewing, 1988, *Radioactive Waste Forms of the Future*, North-Holland.
- Maheras, S. J., A. S. Rood, S. O. Magnuson, M. E. Sussman, and R. N. Bhatt, 1994, *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment*, EGG-WM-8773, EG&G Idaho, Inc., May.

- Martin Marietta Energy Systems, Inc. (Martin Marietta), 1994, EG&G Idaho, Inc., and Westinghouse Savannah River Company, *Radiological Performance Assessment for the E-Area Vaults Disposal Facility*, prepared for Westinghouse Savannah River Company, Aiken, South Carolina, WSRC-RP-94-218, April.
- Smith, T. H., and W. A. Ross, 1975, *Impact Testing of Vitreous Simulated High-Level Waste in Canisters*, Battelle Pacific Northwest Laboratories, BNWL-1903, May.
- Sutter, S. L., 1982, *Accident Generated Particulate Materials and Their Characteristics—A Review of Background Information*, Battelle Pacific Northwest Laboratories, NUREG/CR-2651, May.
- Wood, M. I., R. Khaleel, P. D. Rittman, A. H. Lu, S. H. Finfrock, R. J. Serne, K. J. Cantrell, T. H. DeLorenzo, 1994, *Performance Assessment for the Disposal of Low-Level Waste in the 200 West Area Burial Grounds*, Westinghouse Hanford Company, Richland, Washington, WHC-EP-0645, November.

11. CONCLUSIONS, AND IMPLICATIONS FOR DECISIONMAKING

Section 11.1 presents the key conclusions from this study. Section 11.2 discusses the implications of these conclusions on decisions to be made for managing the INEL ALLW.

11.1 Conclusions

Two release scenarios were evaluated in this study, the undisturbed-waste scenario and the intrusion scenario. The conclusions from the two scenarios are different and are discussed separately below.

The number of PA cases evaluated was quite large (about 120, depending on how one counts) but was still limited. To broaden their usefulness, the conclusions reflect some extrapolations. This allowed including the results expected from cases not specifically evaluated but believed to be predictable with a high degree of confidence, based on the results of closely related cases.

Because of the uncertainties in these evaluations (see Section 13.1), the doses reported herein should be considered approximate. More precise evaluations will be performed later for the particular FWF(s), site, and disposal unit design to be implemented.

11.1.1 Undisturbed-Waste Scenario—Atmospheric Transport Pathway

All FWFs at all sites would be expected to comply with the performance objectives of DOE Order 5820.2A for the undisturbed-waste scenario—atmospheric transport pathway.

- Only long-term release of radon would be expected to result in substantial effects via this pathway.
- To a first approximation, the effects are independent of disposal site.
- For the PCC FWF, the peak outdoor concentration of radon is 0.011 pCi/L. The peak dose is 2.1 mrem/yr.
- The dose would be lower for the other FWFs, which provide more resistance to radon diffusion.
- As discussed in Section 10.3, no standards apply to the flux, concentration, or dose from outdoor releases of radon from the disposed ALLW. However, (a) the peak radon flux from the ALLW is much less than that in the nonapplicable EPA standard, (b) the peak radon concentration is far below the EPA action level for indoor radon, and (c) the peak dose calculated for outdoor radon is about 1% of the 200-mrem/yr average dose for indoor radon in the United States.

11.1.2 Undisturbed-Waste Scenario—Groundwater Transport Pathway

As stated previously, the use of one generic model for the groundwater pathway and the disposal unit design in addressing five sites with widely varying characteristics limits the accuracy to which the current method can predict radionuclide transport and dose via this pathway. Therefore, the conclusions should be viewed as scoping results only and subject to later confirmation by site-specific modeling.

11.1.2.1 Conclusions: Compliance with Performance Objective on Dose. Compliance with the performance objective on dose for the undisturbed-waste scenario—groundwater transport pathway depends on the FWF, disposal site, and overall conservatism of the assumptions made.

Although not every combination of FWF, site, assumptions, and parameter values was evaluated, enough runs were made to develop a semiquantitative display of the conclusions. See Figure 11-1.

The display was developed by examining how the calculated doses in Table 10-2 changed as the various factors were changed. The display is semiquantitative, in that it is useful for making comparisons to within a precision of about a factor of 10. However, it is not a precise numerical plot because it presents a simplified composite of the many factors that were varied. Some of the factors have a nonlinear effect on the calculated dose.

Figure 11-1 is basically a mapping of the dose, in mrem/yr, as a function of the performance of the FWF and the performance of the disposal site. The magnitude of the dose can be imagined as extending out of the plane of the page. The doses are indicated primarily as contour lines of constant dose. Not all calculated doses are shown explicitly because of the impracticality of depicting many data points virtually on top of one another.

The horizontal axis of Figure 11-1 represents the relative performance of the FWF. Moving along the axis toward the left from one division mark to the next represents roughly a tenfold improvement in FWF performance and roughly a tenfold reduction in calculated dose, if all other factors are held constant.

Designators for the six FWFs plus untreated waste (UTR) are located on the horizontal axis so as to depict their effect on dose. It can be seen that the relative performance of IEB4 for this pathway is about $2\frac{1}{2}$ factors of 10 (a factor of about 300) superior to that of UTR. That is, the dose from an IEB4 FWF would be about $1/300$ that for untreated waste, if all other factors are held constant. FWF performance is a function of several variables, but the leach rate is principal among them.

The designator PCC-INT is used on Figure 11-1 to represent the performance of the PCC FWF when chemical interactions between the radionuclides and the cement are considered in the modeling and long-term integrity of the PCC is assumed. Based on the simplified modeling of chemical interactions, the performance of PCC-INT is far better than that of PCC and, at its best, lies between that of IEB and that of IEB4.

Note:
Each mark on vertical and horizontal scales represents roughly a 10-fold effect on dose. Upper end of each vertical bar represents quite conservative assumptions. Lower end represents less conservative assumptions. Numbers on field of figure are calculated values of peak doses (mrem/yr) for combinations of FWF, site, and assumptions and parameter values.
ML5069/27

Figure 11-1. Semiquantitative plot of compliance with performance objective for dose in undisturbed-waste scenario—groundwater transport pathway for Sites A, B, and C.

The vertical axis similarly represents the relative performance of the disposal site. Moving downward along the axis from one division mark to the next represents roughly a tenfold improvement in site performance and roughly a tenfold reduction in dose, if all other factors are held constant.

Performance levels representative of Sites A, B, and C are located on the vertical axis so as to depict their effects on dose. The performance is by far the best for a site exhibiting the characteristics of Site C; the characteristics used for Site B are somewhat more favorable than those used for Site A. Site performance is a function of many variables. Among the most important are the groundwater travel time in the vadose zone and the sorption coefficients.

The performance of each site is shown as a range with a vertical bar, rather than as a single point, for the following reason. Many of the assumptions and parameter values that were varied relate to site characteristics. Examples are K_d values and infiltration rates. Because there is considerable uncertainty in such parameters, even for a given site, a range of values was investigated. One end of the range represents more conservative values; the other, less conservative values. Figure 11-1 accommodates this effect by use of a bar for each site; the length of the bar represents the range of the dose values resulting from differing (but realistic) degrees of conservatism. If truly extreme values—highly conservative or highly nonconservative—were pursued, the bars would be extended correspondingly in both directions.

The performances of Sites D and E are not shown on Figure 11-1. Except for one run involving IEB4 at Site E that barely satisfied the performance objective, all runs involving Sites D and E failed by a very large margin. On Figure 11-1, vertical bars representing Sites D and E would have to be placed approximately 3 factors of 10 (3 division marks) higher than the bar for Site A. The additional vertical bars for Sites D and E would be off the top of the current chart.

The observations that can be drawn from Figure 11-1 and the above discussion include the following:

Sites D and E

- For Sites D and E, only IEB4 at Site E satisfies the performance objective, and that by only a very small margin (20 mrem/yr versus 25 mrem/yr). Any additional conservatism in the assumptions or parameter values would push the resulting dose above the 25-mrem/yr limit.

Sites A, B, and C

- For the remaining site with the least favorable characteristics (Site A) and under very conservative assumptions, the less favorable FWFs (UTR, and PCC and FUETAP without credit taken for chemical interactions with the cement) may or may not ultimately fail the performance objective. The calculated doses are either slightly above the limit of 25 mrem/yr or within a factor of 10 below the limit. The pass-or-fail outcome depends on the conservatism of the specific assumptions used. Even though they pass in the current evaluations, these combinations are susceptible to failing when

the detailed, formal PA is done because the current margin of compliance is smaller than the range of potential conservatism.

- All other combinations of FWF (including PCC and FUETAP with credit taken for chemical interactions in the cement), site, and degree of conservatism satisfy the performance objective. The margin by which these combinations pass ranges from a factor of about 10 to more than a factor of 1,000. Thus, at sites with the characteristics of Site B or C, any waste form would probably satisfy the performance objectives, even untreated waste. The margin would be greater at Site C than at Site B.
- The tradeoff of FWF performance against site performance is evident from the slope of the constant-dose lines in Figure 11-1. An excellent FWF will pass the performance objective at any of the three sites; an excellent site will pass with any FWF that was evaluated. It is noted that, compared with disposal site properties, the properties of FWFs are more readily tailored and more verifiable.

11.1.2.2 Conclusions: Compliance with Performance Objective on Groundwater Protection. Compliance with the performance objective on groundwater protection for the undisturbed-waste scenario—groundwater transport pathway depends on the site and the FWF.

- All FWFs pass at Sites A, B, and C
- All FWFs fail at Sites D and E.

11.1.2.3 Other Conclusions. In addition to the conclusions regarding compliance with the performance objectives, the following general conclusions can be drawn about system performance for the undisturbed-waste scenario—groundwater transport pathway:

- The results depend most strongly on the assumed values for the leach rate of the FWF, the infiltration rate, and the K_d s.
- The results also depend strongly on whether credit is taken for radionuclide sorption in the source zone.
- In most cases, the dominant radionuclide contributing the greatest to the peak dose is the Am-241 decay chain, in particular, Np-237. However, for most runs with Sites D and E and for the most conservative runs at Site A, the U-233 decay chain is the dominant contributor.

11.1.3 Intrusion Scenarios

To a first approximation, the intrusion doses calculated with the present methodology are independent of disposal site. Therefore, the evaluations focused on the FWF and on the assumptions made.

11.1.3.1 Conclusions: Compliance with Performance Objectives on Dose.

Compliance with the performance objectives on dose for the intrusion scenarios depends on the FWF, disposal parameters, and overall conservatism of the assumptions made.

- For the intruder-drilling scenario, all FWFs and burial depths studied satisfy the performance objective, except for cases in which an intruder drills into the RH waste (an extremely unlikely situation) rather than into the CH waste or an assumed homogeneous mixture of the RH and CH waste.
- For the intruder-excavation scenario, all FWFs and burial depths studied satisfy the performance objective. The one exception was the case involving TRU waste, not ALLW.
- For the intruder-agricultural scenario with the 5-ft burial depth (16-ft total cover), all FWFs fail to satisfy the performance objective.
- For the intruder-agricultural scenario with the 9-ft burial depth (20-ft total cover), all FWFs fail if an intruder drills into the RH waste (an extremely unlikely situation) rather than into the CH waste or an assumed homogeneous mixture of the RH and CH waste. If it is assumed that this does not occur, then:
 - The PCC, FUETAP, BSG, ASG, and untreated FWFs pass
 - The IEB FWF passes if (a) the refined inventory or the inventory with additional refinements is used or (b) the refined particle size distribution is used
 - The IEB4 FWF passes if (a) the inventory with additional refinements is used or (b) the refined particle size distribution is used
 - The TRU waste fails.
- For the intruder-radon scenario, all FWFs pass.
- In summary, satisfying the performance objectives for intrusion requires (a) burial at least 9 ft deep (20-ft total cover), (b) special provisions for the RH ALLW, and (c) in the case of the IEB and IEB4 FWFs, either use of the improved inventory or use of the refined particle size distribution in the PA calculations.

11.1.3.2 Other Conclusions. In addition to the conclusions regarding compliance with the performance objectives, the following general conclusions can be drawn about system performance for the intrusion scenarios:

- The results depend particularly on the depth of burial (belowgrade), cover longevity, and radionuclide concentration in the FWF

- Improved immobilization of the radionuclides into a more concentrated FWF can degrade the intrusion performance unless credit is taken for the likely larger particle sizes of the improved FWF when subjected to drilling
- The dominant radionuclides contributing to dose are Cs-137 and Am-241 during the early period studied (100 yr postclosure) and Am-241 and Pu-239 for the later period (500 yr postclosure)
- Separate runs confirmed that the peak doses to intruders are predicted to occur at 100 yr and 500 yr postclosure.

11.2 Implications for Decisionmaking

11.2.1 Introduction

Performance assessment is an important part of decisionmaking for the treatment and disposal of LLW. Selection of a FWF is a major decision involving very large funds, if the cost of implementing the decision is considered (facility construction and operation). If a FWF with inadequate performance is selected and implemented, the cost and schedule impacts from later discovery that the performance objectives are not met could be extremely serious. On the other hand, implementation of a FWF with an unnecessarily high level of performance could lead to excessive expenditures on waste treatment.

This section discusses briefly the implications of the information developed in this preliminary study upon the decisions to be made in the PSPI project. It draws primarily from the conclusions in Section 11.1. It draws also from the later discussions of the conservatism of the present study, found in Section 12, and the uncertainties and areas needing further study, discussed in Section 13.

The PSPI project is still in its early stages. Many future events and external factors may affect the future direction of the project. Therefore, the observations and recommendations presented here are an assessment of only the present situation and may not reflect future conditions.

11.2.2 Observations and Recommendations

The following observations and recommendations are offered, based on the information developed in this preliminary parametric PA:

- PA considerations should continue to be part of the waste treatment project. As the project moves forward and focuses more closely on specific sites and specific FWFs, the PA studies should become more focused. Several topics meriting further study have been identified.
- Disposal of the INEL ALLW at Site D or Site E or other sites with similar characteristics is not expected to satisfy the performance objectives, regardless of the

FWF. However, the uncertainties in the current generic modeling of the groundwater pathway are particularly sizeable at Sites D and E.

- It is expected that disposal of the INEL ALLW at Sites A, B, or C or other sites with similar characteristics can satisfy the performance objectives.
 - Disposal will need to be at least 9 ft below grade.
 - Credit may need to be taken for the chemical interactions in the PCC and FUETAP FWFs to predict satisfactory performance in the undisturbed-waste scenario—groundwater transport pathway.
 - The improved radionuclide inventory or the refined particle size distribution may have to be used for the IEB and IEB4 FWFs to predict satisfactory performance in the intruder-agricultural scenario.
 - Special provisions are expected to be needed for disposal of the remote-handled ALLW. Disposal of such waste in conventional containers in pits is not expected to satisfy the performance objectives. Possible provisions include disposing of such waste by intermediate-depth disposal or at a deep geologic repository.

12. PERSPECTIVES ON THE ASSUMPTIONS AND PARAMETER VALUES USED: HOW CONSERVATIVE?

12.1 Introduction

In this preliminary parametric PA, a major emphasis was placed on investigating the effects that the assumptions and parameter values have on the results. The study was performed in three phases. Results from the first phase guided the selection of assumptions and parameters to be investigated in the second and third phases.

The results from the first phase indicated that the assumptions and parameter values strongly affected the results. In particular, the degree of conservatism in the assumptions and parameter values is important in projecting the acceptability of a FWF. A given FWF at a given disposal site could be projected to either pass or fail the performance objectives, depending simply on the assumptions and parameter values used in the PA.

One cannot predict the degree of conservatism that will be required ultimately by the personnel who review and approve the formal PA addressing disposal of the treated ALLW. If the reviewers require great conservatism, FWFs will tend to be pushed down the scale of acceptability. Some FWFs may fall below the limit of acceptability by failing the performance objectives.

If the assumptions and parameter values used in this study are quite conservative, the chances are reduced that future reviewers will push FWFs below the levels of acceptability arrived at herein. On the other hand, if this study is excessively conservative, attractive FWFs may be prematurely ruled out.

The question of the degree of conservatism of this study is, therefore, of considerable importance. That question was addressed (as explained in Section 12.2) by comparing the assumptions and parameter values used herein against those in five other PAs or risk assessments related to the SDA (Site A).

12.2 Comparisons against the Assumptions and Parameter Values Used in Related Assessments

Hundreds of assumptions and parameter values are used in a PA. Assumptions are necessary where data are unavailable. Assumptions are also necessary in cases where complex natural processes need to be simplified in order to allow simulation with computer models. Single values of parameters need to be selected from a range of possible and reasonable values. Making assumptions and selecting parameter values are intimately associated with the resulting uncertainties in the results of a PA. Only limited, broad direction (e.g., DOE 1994) is available on the assumptions and parameter values to be used in PAs prepared in response to DOE Order 5820.2A.

From among the hundreds of assumptions and parameter values used in a PA, a panel of technical personnel selected approximately 130 expected to be among those having the greatest effect on the calculated doses.

For these approximately 130 parameters, the assumptions and values used in this study were compared systematically against those in five related PAs or risk assessments. The assessments are related in that they all dealt with various portions of the waste that either are now, or may be in the future, disposed of in the SDA (Site A). The assessments differed, however, in the specific waste addressed, purpose, level of detail, number of alternatives evaluated, and intended degree of conservatism. It is understandable, therefore, that the studies used a range of assumptions and parameter values.

The five other assessments are as follows:

- The formal RWMC Performance Assessment (Maheras et al. 1994), which addresses the LLW buried in the SDA from 1984 through 2020
- The Preliminary Scoping Risk Assessment (Loehr et al. 1994), prepared pursuant to CERCLA, which addresses both radiological and nonradiological contaminants in the TRU waste and LLW buried in the SDA from 1952 through 1983
- An informal PA (Rood 1992) on the waste buried in the SDA from 1954 through 1970, which was assumed to be retrieved, treated to produce an IEB FWF, and buried in the SDA
- An informal PA (Maheras et al. 1992) addressing the same situation as that immediately above, but with a concrete FWF
- An informal PA (Rood 1993) on the INEL ALLW (without the nonactinides), which was assumed to be disposed of in the SDA without treatment.

The last three assessments listed were less detailed than the first two.

The primary purpose of the comparisons was to assess the degree of conservatism in the first phase of this study, compared with that in the other five assessments, thereby identifying any additional computer runs needed for the second and third phases of this study. An additional purpose was to gain a better understanding of the important processes and parameters that control disposal system performance. A final benefit of the comparisons is that the resulting compilation may be helpful in preparing future PAs, risk assessments, and other environmental documents related to the SDA.

12.3 Results of the Comparisons

The detailed comparisons appear in Appendix H.

Based on the results of the comparisons, additional computer runs were made in the second and third phases of this study to evaluate the effects of alternative assumptions or different parameter values. The new computer runs are discussed in Section 10. The comparisons in Appendix H were then updated to reflect the new computer runs.

Table 12-1 is a simplified summary of Appendix H. The first column lists the most important of the approximately 130 parameters covered in the comparisons. As discussed in Appendix H, "important" parameters are defined here as those for which (a) the values assumed in the six assessments had a large spread and (b) a substantial effect on the results was expected. The second column provides a simplified conclusion on whether the first-phase evaluations herein were in the conservative or nonconservative part of the spectrum of reasonable values. The last column of Table 12-1 discusses any new computer runs made and/or indicates the degree of conservatism after the second and third phases of the study.

Table 12-1 reflects a great simplification of Appendix H. For details, the latter should be consulted.

12.4 Conclusions

The degree of conservatism of a PA is difficult to assess in quantitative terms. In the present situation, even though six assessments addressed the same disposal area, many aspects of the assessments were so different that no rigorous conclusions can be drawn. Short of repeating the entire PA with the exact assumptions and parameter values used in other assessments, there is no way to give a quantitative answer to the question, "How conservative is this PA?"

Nevertheless, the comparisons performed here do provide a perspective on the degree of conservatism in this study. For the undisturbed-waste scenario, the first-phase evaluations tended to be conservative on the points of taking no credit for waste containers or for radionuclide sorption in the source zone. The evaluations tended to be nonconservative on some of the assumed infiltration rates and on the vadose zone travel time. These nonconservatisms were investigated in the second phase by making computer runs with more conservative assumptions and parameter values. Overall, the effects of the conservative vs. nonconservative assumptions tended to balance each other. For Site A, comparing the calculated dose for the second-phase Run 1IR2NBSZS with that for Run 1 shows that the effects almost exactly cancel out. (Run 1 used the lower infiltration rate, took credit for the basalt layers, but took no credit for sorption in the source zone. Run 1IR2NBSZS used the opposite assumptions—the higher infiltration rate, no credit for the basalt layers, but credit for sorption in the source zone.) The combined set of computer runs for the two phases provided relatively complete coverage for the important parameters.

For the intruder scenarios, the first-phase computer runs tended to be conservative, principally because of the assumption that spread the nonactinide inventory over all of the waste. The other assumptions tended to balance out one another, being comparatively conservative on some points and comparatively nonconservative on others. The second- and third-phase computer runs provided refined assumptions about the nonactinide distribution and reduced the conservatism in some cases.

The overall conclusion is that this study encompasses a suitable range of conservatism in the various computer runs. It seems unlikely that the required conservatism for the formal PA addressing disposal of the ALLW will fall outside the range of conservatism investigated here.

Table 12-1. Degree of conservatism of the most important assumptions and parameter values used in this study.

Assumption or parameter	Degree of conservatism in first-phase evaluations	Degree of conservatism after second- and third-phase evaluations
Split of CH and RH waste	Conservative	Refined analyses produced lower doses in some cases and showed that first-phase computer runs for intrusion scenarios were very conservative for CH waste
Degradation of FWF	Conservative	Covered full range of assumptions
Credit for waste containers	Conservative	Conservative
Credit for contaminant adsorption in cement	Conservative	Covered full range of assumptions
Burial depth	Typical	Typical
Cover integrity	Typical	Typical
Comprehensiveness of list of transport pathways and exposure routes for undisturbed-waste scenario	Less conservative, but effect on results is small	Less conservative, but effect on results is small
Infiltration rate	Nonconservative	Covered full range of values
Credit for cover reducing infiltration during institutional control	Conservative	Conservative
Vadose zone travel time	Nonconservative	Covered full range of assumptions
Flooding scenario	Typical but nonconservative	Typical but nonconservative
Sorption in waste zone	Conservative	Covered full range of assumptions
Transport pathways and exposure routes for intrusion scenarios	Approximately in middle of range	Approximately in middle of range
Assumptions governing exposure during intruder-drilling scenario	Nonconservative	Covered full range of assumptions

Table 12-1. (continued).

Assumption or parameter	Degree of conservatism in first-phase evaluations	Degree of conservatism after second- and third-phase evaluations
Assumptions governing exposure during intruder-excavation scenario	Conservative	Conservative
Assumptions governing exposure during intruder-agricultural scenario	Approximately in middle of range	Approximately in middle of range

References for Section 12

- DOE, 1994, *Performance Assessment Task Team Progress Report*, U.S. Department of Energy, DOE/LLW-157, Rev. 1, May.
- Loehr, C. A., B. H. Becker, D. E. Burns, R. M. Huntley, S. M. Rood, P. Sinton, and T. H. Smith, 1994, *Preliminary Scoping Risk Assessment for Waste Pits, Trenches, and Soil Vaults at the Subsurface Disposal Area, Idaho National Engineering Laboratory*, EG&G Idaho, Inc., EGG-WM-11181, May.
- Maheras, S. J., A. S. Rood, and R. Shuman, 1992, "A Performance Assessment of Radioactive Waste Encapsulated in Concrete and Disposed at the INEL," September (located in EG&G Idaho, Inc., Engineering Design File IWPF-0028, Appendix 2).
- Maheras, S. J., A. S. Rood, S. O. Magnuson, M. E. Sussman, and R. N. Bhatt, 1994, *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment*, EG&G Idaho, Inc., EGG-WM-8773, May.
- Rood, A. S., 1992, "A Performance Assessment of Radioactive Waste Encapsulated in Iron-Enriched Basalt and Disposed at the Idaho National Engineering Laboratory's Radioactive Waste Management Complex", *Proceedings of the International Topical Meeting on Nuclear and Hazardous Waste Management, Spectrum '92*, August (located in EG&G Idaho, Inc., Engineering Design File IWPF-0028, Appendix 3).
- Rood, A. S., 1993, "A Limited Performance Assessment for the Idaho Waste Processing Facility: 1993 Update Using Revised Parameters and Limited Sensitivity Analysis", September 29, (located in EG&G Idaho, Inc., Engineering Design File IWPF-0044).

13. UNCERTAINTIES AND AREAS NEEDING FURTHER STUDY

13.1 Uncertainties

The identification and discussion of the uncertainties are important parts of a PA. The uncertainties include those related to the major areas of a PA: the radionuclide inventory, the release mechanisms and rates, the physical setting of the site, the scenarios, the conceptual and numerical models used to estimate transport, and the exposure parameters. The uncertainties can be classified into three broad categories (EPA 1994): scenario uncertainties, model uncertainties, and parameter uncertainties.

The variables and assumptions believed to contribute most of the uncertainty in the current study were identified. Most of the emphasis was placed on the modeling uncertainties and the parameter uncertainties. No attempt was made to list every uncertainty or to quantify precisely the total degree of uncertainty in the PA results.

Table 13-1 summarizes the key uncertainties. The potential effect of each entry is indicated in terms of possibly overestimating or underestimating the dose. Steps taken to investigate or reduce the uncertainty are listed. For example, many of the key uncertain parameters were varied over a range of values to determine the effect on the calculated dose. Finally, comments and conclusions are provided as appropriate.

13.2 Areas Needing Further Study

This section determines the areas listed in Table 13-1 and elsewhere for which further study is recommended at this time.

13.2.1 Areas that Are Sufficiently Well Defined Programmatically to be Considered for Further Study

Some areas related to the disposal of the INEL ALLW are fairly well defined programmatically at this time. Other areas are not yet well defined. Pursuing technical uncertainties in the latter may not be cost effective. This section distinguishes between the two types of areas.

The first paragraph in Section 13.1 listed the major areas of a PA. In an overview sense, the programmatic status of those areas is as follows:

- Radionuclide inventory

This area is comparatively well defined, in that the body of waste to be addressed has been identified. The programmatic uncertainties, such as (a) the lower limit of TRU concentration for ALLW, and (b) the exact details for dividing containers between those that are TRU waste and those that are ALLW, are not expected to cause major impacts on the PA.

Table 13-1. Key uncertainties in this study.

Uncertainty factor	Potential effect of uncertainty on calculated dose	Steps taken to investigate or reduce uncertainty	Comments/conclusions
The activities of the radionuclides in the ALLW are uncertain, due to limited data on the waste, uncertainties in the partitioning between TRU waste and ALLW, and unknowns about the effects of treatment.	May overestimate or slightly underestimate dose.	Used conservative upper limit of 100 nCi/g TRU for most computer runs. Performed computer runs with 10 and 1,000 nCi/g TRU to observe effects. Used conservative values for activity of nonactinides, in particular Cs-137, which was found to dominate intruder dose in many cases. Examined effect of separating vs. combining CH and RH waste in the model. Examined effect on dose if activity of difficult-to-measure radionuclides increased.	This uncertainty is an important factor. It was investigated rather thoroughly here. The range of values used is believed to bound the actual values.
Very-small-activity radionuclides were screened out of the evaluations.	May slightly underestimate dose.	Subsequent runs verified that the screened-out radionuclides had a negligible effect on the dose.	The uncertainty is quite small.
The modeling of radionuclide release mechanisms and release rates was greatly simplified.	May overestimate or underestimate dose.	Performed computer runs with maximum and minimum release parameter values for each category of FWF (cement, ceramic, glass). Performed runs with fractured FWF. Performed runs on untreated waste as a benchmark for comparison. Took no credit for waste container.	This uncertainty is an important factor. It was investigated rather thoroughly here. The range of values used is believed to bound the actual values.
No credit was taken for sorption of contaminants in cement FWFs in most runs.	May overestimate dose.	Performed computer runs in which sorption in the cement was modeled using a simple method.	The uncertainty is an important factor. The range of assumptions used is believed to bound the actual values.
The timeline modeled for ALLW treatment, disposal, and facility closure was simplified by using an unrealistically short interval (1 yr).	May overestimate dose.	None.	This uncertainty is believed to be rather small. In most cases, the dose is dominated by the long-lived actinide decay chains, and a delay of up to 20 or 30 yr in the time period modeled would have a negligible effect. For many intrusion scenarios, however, the dose is dominated by Cs-137. The same delay of 20 or 30 yr could result in a reduction of as much as 50% in the Cs-137 dose, due to radioactive decay occurring before closure of the disposal unit.

Table 13-1. (continued).

Uncertainty factor	Potential effect of uncertainty on calculated dose	Steps taken to investigate or reduce uncertainty	Comments/conclusions
The location of the disposal site is unknown.	May overestimate or underestimate dose.	Performed computer runs for various FWFs at five possible disposal sites with large differences in characteristics.	This uncertainty is an important factor. The range of sites studied is believed to provide a broad perspective on the range of potential behaviors of the site that could conceivably be selected.
The disposal unit layout (burial depth, backfill thickness, engineered barriers, cover) is uncertain.	May overestimate or underestimate dose.	Performed computer runs with different depths to top of waste stack. Assumed a well-engineered cover, which retains integrity for its design life, then instantly erodes totally. In most runs, took no credit for engineered barriers. Took no credit for sediment underlayer in pit.	The depth to the top of the waste stack is an important factor for the intrusion scenarios. The range of values used is believed to bound the actual values. The other disposal unit variables are believed to be less important.
Natural disturbances of the waste zone (e.g., by flooding) were not evaluated.	May underestimate dose.	None.	The effects of natural disturbances should be evaluated when specific candidate sites have been identified formally.
The infiltration rate for precipitation at the disposal units is uncertain.	May underestimate dose for some computer runs.	Infiltration rate for most computer runs is probably nonconservative. Performed several runs with larger infiltration rates, up to a large upper limit.	This uncertainty is an important factor. The range of values investigated is believed to bound the actual values.
The modeling of the subsurface structure, flow, and transport was greatly simplified.	May overestimate or underestimate dose.	Performed computer runs to investigate effects of some of the simplifications. Examples are effects of taking no credit for basalt layers at Site A, effects of taking no credit for radionuclide sorption in the waste zone.	The scope of the study did not allow detailed site-specific modeling. This uncertainty is an important factor.
The values of sorption coefficients are uncertain.	May overestimate or underestimate dose.	Used representative values for most of the computer runs. Performed runs with more conservative values, also.	This uncertainty is an important factor. It was investigated rather thoroughly here, at least for the present level of detail of the study. The range of values used is believed to bound the actual values.

Table 13-1. (continued).

Uncertainty factor	Potential effect of uncertainty on calculated dose	Steps taken to investigate or reduce uncertainty	Comments/conclusions
For groundwater transport, it was assumed that all progeny arrive at the receptor well at the same time as the parent.	May overestimate or underestimate dose.	None.	In reality, a chromatographic effect will occur, with progeny arriving at the receptor well at different times. This spreading of the arrival times will tend to reduce the peak dose. On the other hand, progeny with high dose conversion factors and fast travel times are retarded so as to travel with the parent in the model, which may tend to underestimate dose.
For the undisturbed-waste scenario, only the groundwater transport pathway and radon transport via the airborne pathway were evaluated.	May underestimate dose.	None.	Results for other pathways from Maheras et al. (1994) were used to justify not evaluating the dose from the other pathways.
The exposure routes and receptor locations are uncertain.	May overestimate or underestimate dose.	Performed computer runs using generally conservative exposure routes and conservative values of the related parameters. Performed one intrusion run using a more conservative scenario for drilling.	The assumptions used are somewhat conservative. If direct intrusion into the waste does not occur, the dose will have been overestimated considerably.
The dose conversion factors are uncertain.	May overestimate or underestimate dose.	None.	Widely accepted values were used.

- Release mechanisms and rates

This area depends on the FWF(s) that will be selected and is thus not well defined. However, the FWF(s) selected will most likely represent one or more of the three major categories evaluated here: cement, glass, ceramic. The range of specific formulations for each category was substantially bounded by the choice of the six FWFs studied here.

- Physical setting of site

A formal list of candidate sites, from which the actual site will certainly be selected, has not been prepared. Thus, this area is not well defined. Pursuing site-specific details (e.g., K_d s, infiltration rates), without assurance that the particular site is a formal candidate, may not be cost effective.

- Scenarios

Scenarios for PA evaluations are generally independent of programmatic considerations. Some aspects of scenarios are site-specific, however, and therefore not well defined programmatically.

- Conceptual and numerical models used to estimate transport

To a large extent, detailed models of radionuclide transport are site specific.

- Exposure parameters

Some exposure parameters are site specific.

Based on the above arguments, the areas that tend to be the best suited for additional study at this time are related to the radionuclide inventory and the release rates. The areas that tend to be the least suitable are any involving site-specific parameters.

13.2.2 Areas for Further Study

After consideration of the technical uncertainties from Table 13-1 and the program status from Section 13.2.1, the following areas are recommended for further study at this time.

- Better information is needed on the operational approach for dealing with CH vs. RH ALLW in treatment and disposal. In addition, a more precise estimate of the TRU content in the treated ALLW is needed to supplement the upper and lower limits used herein. Both of these items would be addressed by examining possible operational schemes for treatment and disposal, then considering the impacts on the radionuclide inventory in the treated ALLW.
- The simplified timeline used in this study (which assumed only one year for waste treatment, disposal, and site closure) represents a worst-case situation and is not realistic.

Some computer runs should be made using a timeline that represents the current best judgment for the programmatic and operational timeline.

- Several simplifications were made in modeling the release mechanisms; the subject needs to be addressed in more detail as the selection of a FWF becomes better defined.
- This preliminary parametric PA produced an initial mapping of the projected compliance status of various combinations of FWFs and disposal sites, for various degrees of conservatism in the analysis (see Section 11). The mapping is not precise; it is only semiquantitative. It is intended to indicate the general behavior of the "compliance versus noncompliance" conclusion in the complex, multivariable space of possible disposal system combinations. The mapping is based on the results of the limited number of cases evaluated. The limited number of cases necessitated some extrapolation of results for cases that were not evaluated but were estimated based on results of similar cases. Additional cases need to be evaluated to provide a more detailed and defensible mapping.

References for Section 13

EPA, 1994, *Exposure Assessment Guidelines*, U.S. Environmental Protection Agency, EH-232-009/08/94, August.

Maheras, S. J., A. S. Rood, S. O. Magnuson, M. E. Sussman, and R. N. Bhatt, 1994, *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment*, EG&G Idaho Inc., EGG-WM-8773, May.

Appendix A

Estimating the Radionuclide Inventory in INEL-Stored Alpha Low-Level Waste

Appendix A

Estimating the Radionuclide Inventory in INEL-Stored Alpha Low-Level Waste

Incomplete information is available from shipping records concerning the radionuclide inventory in the alpha low-level waste (ALLW) stored at the Idaho National Engineering Laboratory (INEL). Therefore, that information was supplemented with assumptions, analyses, and other sources of information to arrive at a suitable inventory for use in this study.

This appendix describes how the radionuclide inventory was estimated. The inventory is in terms of the activity (in Ci) and the volumetric concentration of activity (in Ci/m³) of each radionuclide.

Three successive approximations of the radionuclide inventory were developed and used for runs involving treated ALLW. Each approximation was more detailed and is expected to be more accurate than its predecessor.

The first version of the inventory is called the "simplified inventory." It was developed by means of the simplified approach described in Section A-1 through A-8. It was used for all of the computer runs in the first phase of the study and for most of the runs in the second and third phases. Unless otherwise stated, this was the inventory used.

For the second phase, a "refined inventory" (described in Section A-9) was developed and used for a limited number of computer runs. Those runs are so identified in the presentation of results.

For the third phase, an "inventory with additional refinements" (described in Section A-10) was developed and used for a limited number of computer runs. Those runs are so identified in the presentation of results.

Appendix B describes the refined model of the untreated ALLW used for a limited number of computer runs in the second phase of the study.

A-1. BACKGROUND

The posttreatment inventory of radionuclides in the ALLW cannot be estimated with a high degree of confidence before the treatment campaign is well underway. Several reasons exist for this uncertainty:

- The shipping records often did not provide complete lists of the radionuclides in the waste
- Determination of the plutonium in the waste was difficult because of measurement limitations

- Measurement of some nonactinides in the waste was also difficult because the radionuclides are β -emitters that have no associated γ radiation (e.g., Sr-90, Ni-63)
- In addition, the outcome of dividing the waste into waste that is >100 nCi/g after treatment and waste that is ≤ 100 nCi/g after treatment is difficult to predict.

Thus, rigorous projection of the inventory of the radionuclides in the posttreatment waste is not possible.

In the absence of data to support rigorous inventory estimates, limiting or best-judgment inventories were developed based on several assumptions and simplifications. The result of the assumptions and simplifications was a simplified inventory (with later refinements) that allowed the performance assessment (PA) to proceed cost-effectively with the assurance that realistic, if not precise, values were used for the inventory.

Sections A-2 through A-8 discuss the basis for the simplified inventory and present the result.

A-2. CONSTRAINTS

The radionuclide inventory had to satisfy several constraints to be comprehensive and realistic:

- It had to include only the ALLW, not the transuranic (TRU) waste
- It had to include the waste from all generators, not just Rocky Flats Plant (RFP), which was the predominant generator
- It had to include all of the radionuclides in the waste, not just the actinides (the radionuclides with atomic number ≥ 89 and their progeny are referred to here as the "actinides"; all other radionuclides are referred to here as the "nonactinides")
- It had to represent posttreatment conditions, not as-stored conditions in the Transuranic Storage Area (TSA) at the Radioactive Waste Management Complex (RWMC)
- It had to address the three broad categories (glass, ceramic, cement) of final waste forms (FWFs) under consideration, including the two extreme or limiting FWFs (more favorable and less favorable) for each category
- The list of radionuclides had to be complete enough to represent essentially all of the risk, but short enough to allow cost-effective, repeated parametric calculations of radionuclide transport.

The inventory of radionuclides in the ALLW has significant uncertainties. Therefore, for the actinides, upper-limit and lower-limit inventories were developed to bound the actual inventory. For the nonactinides, the inventory is a best estimate.

A-3. GENERAL ASSUMPTIONS

Three very broad, simplifying assumptions were made:

- In addressing each of the six FWFs (one at a time), all waste is in the same FWF
- Within a given FWF, the radionuclide concentrations are homogeneous over all the posttreatment waste
- All radionuclides that enter treatment with the waste stay with the treated waste.

A-4. ACTINIDES

A-4.1 Initial Steps

The inventory of actinides was developed as explained below. As will be seen, the inventory of actinides used in most of the computer runs is an upper-limit inventory. Table A-1 shows the various steps in the process.

The starting point was the pretreatment inventory for all waste emplaced in the TSA through 1992, as found in the Radioactive Waste Management Information System (RWMIS) database of shipping records. The 1992 data were the most recent available at the time this study began. Only a very few drums of waste have been emplaced since 1992.

The first column of Table A-1 lists the radionuclides in the inventory that are actinides.

Some of the actinides fall within the definition of TRU radionuclides given in DOE Order 5820.2A: atomic number >92, alpha emitter, half-life >20 years. Some do not. See the second column of Table A-1. This division of actinides into TRU and non-TRU radionuclides is key to the approach used here.

Atwood and Schlafman (1993) decayed the RWMIS inventory data discussed above from the time of emplacement to 1993. The decayed values appear in the third column of Table A-1.

A-4.2 Screening

The inventory in the third column was subjected to three screening steps to reduce the list of radionuclides to only those of potential importance. As a first step, radionuclides in the third column with <0.1 Ci activity were deleted. This step eliminated Ac-227, Am-242m, Bi-214, Bk-249, Cf-249, Cf-252, Cm-242, Cm-243, Cm-246, Cm-248, Pa-233, Pb-210, Pb-214, Po-214, Po-218, Pu-236, Ra-226, Ra-228, Rn-222, Th-230, Th-234, U-228, U-235, and U-236.

The next screening step was to delete the relatively small activities of eight progeny that had grown in during the average period of 15 years since the time of waste emplacement. The maximum activity of any of these progeny in 1993 was 26.55 Ci in the TRU waste plus the

Table A-1. Development of simplified inventory of actinides.

Radionuclide	TRU radionuclide? ^a	Inventory in TRU waste and ALLW in 1993 (Ci)	Screened inventory (Ci)	Inventory normalized to 1 Ci total for TRU radionuclides ^b	Inventory scaled up to 100 nCi/g (after treatment) in all ALLW (Ci)
Ac-227	N	0.04	—	—	—
Am-241	Y	89,780	89,780	0.448	1,340
Am-242m	Y	0.0003	—	—	—
Am-243	Y	0.38	—	—	—
Bi-212	N	26.55	—	—	—
Bi-214	N	<E-04	—	—	—
Bk-249	N	0.0006	—	—	—
Cf-249	Y	0.01	—	—	—
Cf-252	N	0.004	—	—	—
Cm-242	N	<E-04	—	—	—
Cm-243	Y	0.015	—	—	—
Cm-244	N	539	539	0.0027	8
Cm-246	Y	0.002	—	—	—
Cm-248	Y	<E-04	—	—	—
Np-237	Y	0.3552	—	—	—
Pa-233	N	<E-04	—	—	—
Pb-210	N	0.018	—	—	—
Pb-212	N	26.55	—	—	—
Pb-214	N	<E-04	—	—	—
Po-212	N	17.01	—	—	—
Po-214	N	<E-04	—	—	—
Po-216	N	26.55	—	—	—
Po-218	N	<E-04	—	—	—
Pu-236	N	0.015	—	—	—
Pu-238	Y	60,800	60,800	0.303	906
Pu-239	Y	40,150	40,150	0.200	598
Pu-240	Y	9,825	9,825	0.049	147
Pu-241	N	161,100	161,100	0.803	2,401
Pu-242	Y	0.9451	—	—	—
Ra-224	N	26.55	—	—	—
Ra-226	N	0.048	—	—	—
Ra-228	N	0.03	—	—	—
Rn-220	N	26.55	—	—	—
Rn-222	N	<E-04	—	—	—
Th-228	N	26.55	—	—	—
Th-230	N	0.02	—	—	—
Th-232	N	0.3301	—	—	—
Th-234	N	<E-04	—	—	—
Tl-208	N	9.54	—	—	—
U-228	N	<E-04	—	—	—

Table A-1. (continued).

Radionuclide	TRU radionuclide? ^a	Inventory in TRU waste and ALLW in 1993 (Ci)	Screened inventory (Ci)	Inventory normalized to 1 Ci total for TRU radionuclides ^b	Inventory scaled up to 100 nCi/g (after treatment) in all ALLW (Ci)
U-232	N	26.04	—	—	—
U-233	N	898.5	898.5	0.0045	13
U-234	N	5.778	—	—	—
U-235	N	0.067	—	—	—
U-236	N	0.001	—	—	—
U-238	N	0.1171	0.1171	5.8E-7	0.0017
Total		363,313	363,093	NA	5,413

a. A TRU radionuclide is one with an atomic number greater than 92. The definition of TRU waste in DOE Order 5820.2A includes only TRU radionuclides that are alpha-emitters and have half-lives greater than 20 years. The additional stipulations in the DOE Order are reflected in the designation of TRU radionuclides in this table.

b. All entries in the preceding column were multiplied by the normalization constant of 4.986 E-06 to arrive at the entries in this column.

Note: Totals may not match exactly because of rounding.

ALLW. As will be seen shortly in the description of the scaling process, the 26.55 Ci corresponds to an activity of <0.4 Ci for any of these progeny in the ALLW in 1993. (The performance assessment calculations will allow the ingrowth again of these same radionuclides. In a period of about 15 years, essentially the same activities of the progeny will reappear.) This step results in the temporary elimination of progeny Bi-212, Pb-212, Po-212, Po-216, Ra-224, Rn-220, Th-228, and Tl-208, which were not present in the original inventory at the time of shipment.

The last screening step was to delete several additional radionuclides. After the scaling steps to be discussed shortly, the activities of Am-243, Np-237, Pu-242, Th-232, and U-234 would all be <0.1 Ci. The activity of U-232 would be <0.4 Ci. Therefore, these radionuclides were eliminated from further analysis.

The result of these screening steps was to reduce the number of actinides to be modeled from 46 to 8. See the fourth column of Table A-1. The screening process decreased the total activity by less than 0.1%.

A-4.3 Scaling Preliminaries

Based on the screened inventory for actinides in the TRU waste plus ALLW, simplified upper-limit and lower-limit inventories were estimated for the ALLW alone by use of a scaling process. Unless indicated otherwise, all of the computer runs were made using the upper-limit inventory of actinides.

The scaling process was rather involved. The first step was to assume that the activities of the respective actinides in the ALLW have the same ratios to one another as they do in the TSA waste taken as a whole (ALLW plus TRU waste). The justification for this assumption is that the TSA waste generally derives from the same processes that deal with weapons-grade plutonium or heat-source plutonium, whether the waste is the higher-concentration TRU waste (>100 nCi/g) or the lower-concentration ALLW (≤ 100 nCi/g).

With the above assumption, the screened inventory in the fourth column could then be scaled to produce limiting inventories of actinides in the ALLW. The largest potential inventory of actinides in the ALLW corresponds to the situation in which every package of treated waste contains 100 nCi/g of TRU radionuclides. (If the concentration were larger than 100 nCi/g of TRU radionuclides, the waste would be TRU waste, not ALLW.) Correspondingly, the smallest potential inventory of actinides in the ALLW corresponds to the situation in which every package of treated waste contains 10 nCi/g of TRU radionuclides. (If the concentration were smaller than 10 nCi/g of TRU radionuclides, the waste would be non-alpha LLW, not ALLW.) These two limiting cases represent the upper and lower limits on the actinide inventory in the treated waste.

One general assumption stated in Section A-3 was that all radionuclides that enter treatment with the waste stay with the treated waste. Therefore, the pretreatment inventory, in Ci, can be used for the posttreatment inventory.

In preparation for scaling the screened actinide inventory to the limiting cases, the inventory was first normalized to a total of 1 Ci of TRU radionuclides. All entries were multiplied by the normalization constant of $4.986\text{E-}06$. The constant is the inverse of the total activity of long-lived, alpha-emitting transuranics in the fourth column. The result appears in the fifth column of Table A-1.

A-4.4 Scaling, with the Effects of One Method of Treatment

The upper-limit actinide inventory was developed for the situation in which all containers of treated waste have 100 nCi/g of TRU radionuclides. In this situation, the calculated inventory depends on the total mass of the treated waste. (The mass of the empty containers is not considered in this analysis.) To complicate things, the posttreatment mass (a) differs from the pretreatment mass and (b) depends on the treatment method.

The estimated pretreatment mass is based on a waste volume of $24,413\text{ m}^3$ for the ALLW (Raivo, Eddy, and Anderson 1993) and an average waste density of 652 kg/m^3 from evaluations conducted in the INEL Idaho Waste Processing Facility (IWPF) Project. The result is an estimated pretreatment mass of $1.59\text{E}+10$ g.

For simplicity in examining the effects on activity concentration, waste treatment can be considered to consist of two types of operations. The first type of operation includes those that reduce the mass of the waste. For example, incineration reduces the mass of the combustible portion of the waste. However, the estimated fraction of the ALLW that is combustible is a maximum of only 25–30% (by mass), per the IWPF Project. Suppose that 25% of the ALLW mass were essentially eliminated (reduced to a negligible mass of ash), as a result of incineration, offgassing, etc. Each pretreatment unit of mass would therefore be reduced to approximately 0.75 unit mass.

The second type of treatment covers operations that increase the mass of the waste. Immobilization increases the mass, by the addition of immobilization agents. For the FWFs considered, waste loadings range from 30–60% by mass.

It was calculated that, for the FWFs considered, the midpoint of the extreme values of the total posttreatment mass of the ALLW corresponds to a 40% waste loading. (Posttreatment mass is proportional to the inverse of the waste loading.) Because it was the midpoint, this value was used as a baseline for calculational purposes. (*This midpoint loading happens to correspond to the waste loading for cement, both Portland cement/concrete (PCC) and formed under elevated temperature and pressure (FUEAP). The effects for other FWFs are discussed in Section A-8, after the nonactinide inventory is developed.*) Thus, in this case, each unit mass of postincineration ALLW that undergoes (cement) immobilization increases to about 2.5 unit mass of FWF.

The cumulative effect of the two types of processes is an increase in mass from 1 unit mass of waste before treatment to approximately $0.75 \times 2.5 = 1.88$ unit mass of FWF after treatment. The posttreatment mass is therefore approximately $1.88 \times 1.59\text{E}+10 = 2.99\text{E}+10$ g.

At the upper-limit concentration of 100 nCi/g, the $2.99\text{E}+10$ g has a total TRU radionuclide activity of $100 \text{ nCi/g} \times 2.99\text{E}+10 \text{ g} \times \text{E}-09 \text{ Ci/nCi} = 2,990 \text{ Ci}$. Therefore, the normalized entries in the fifth column of Table A-1 (totaling 1 Ci of TRU radionuclides) were scaled by multiplying by 2,990. The result is the estimated posttreatment inventory of actinides for the ALLW at 100 nCi/g, as indicated in the last column.

A-5. NONACTINIDES

For the nonactinides, the starting point for the inventory calculations was again the RWMIS data, as decayed by Atwood and Schlafman (1993). In this case, a best estimate of the nonactinide inventory was developed, because there was no basis for estimating upper-limit and lower-limit inventories.

A-5.1 Basis

An estimate had to be made about the location of the nonactinides in the TRU waste versus in the ALLW, and in the contact-handled (CH) waste versus in the remote-handled (RH) waste. No exact analysis of the division of these radionuclides is possible at present, so assumptions and simplifications were necessary again.

Almost all of the computer runs were made using a nonactinide inventory based on the following simplified analysis. A few runs in the second phase of the study were made with a refined inventory of nonactinides, described in Section A-9 of this appendix, that are more realistic. A few runs in the third phase were made with a nonactinide inventory with additional refinements, described in Section A-10 of this appendix.

In the simplified inventory it was assumed that, before waste treatment, 100% of the nonactinides at the TSA are located in the very small volumetric fraction of the TRU waste and ALLW that is RH.

For this untreated RH waste, it was assumed that the activity fraction of the nonactinide inventory that is in the ALLW is the same as the volume fraction of waste that is ALLW. That is, it was assumed that the concentration of nonactinide activity is independent of the concentration of TRU activity.

RWMIS indicates that about 41% by volume of the RH waste is believed to contain <100 nCi/g of TRU radionuclides and would therefore be classified as ALLW. Other studies and estimates give values ranging from 27 to 70%. For this simplified calculation, the value was assumed to be 50% ($\pm 20\%$). Since the activity fraction was assumed to equal the volume fraction, 50% of the nonactinide activity was assumed to be in the RH ALLW.

It was assumed in this simplified approach that the nonactinide activity in the RH ALLW waste was spread uniformly throughout all of the posttreatment (RH and CH) ALLW.

Applying these assumptions led to the development of the inventory of nonactinides shown in Table A-2. The first two columns show the TRU waste plus ALLW inventory of nonactinides, as decayed by Atwood and Schlafman (1993) from the time of emplacement to 1993. That inventory was multiplied by 0.5 to give the ALLW nonactinide inventory in the third column.

A-5.2 Screening

The inventory in the third column was subjected to screening to reduce the number of radionuclides to only those of potential importance. First, all radionuclides with <1 Ci of activity were deleted. C-14, Co-58, Cr-51, Eu-150, Eu-152, Eu-154, Eu-155, Fe-55, Fe-59, H-3, Mn-54, Nb-95, Ta-182, Tc-99, Zn-65, and Zr-95 were thereby eliminated. Then several radionuclides were deleted based on application of the screening method of Seitz and Kocher (1993) for the groundwater pathway and on computer runs for the intrusion scenarios. This step resulted in eliminating Ce-144, Cs-134, Kr-85, Ni-63, Pm-147, and Pr-144. The screened inventory appears in the fourth column of Table A-2 and has approximately 98% of the total activity listed in the unscreened inventory.

Two final steps were entailed in developing the nonactinide inventory. One step was to apportion the mixed fission products (MFP) and unidentified β - γ entries, both of which were taken to be 50% Sr-90 and 50% Cs-137, and add the resulting activity to the respective radionuclides indicated. The extremely small activity (0.64 Ci) of mixed activation products was neglected with respect to the much larger activity of Co-60, its most likely predominant constituent. The other step was to adjust the Ba-137m activity to match that of Cs-137, and the

Y-90 activity to match that of Sr-90, based on decay equilibrium considerations. The result is in the last column of Table A-2.

A-6. TOTAL INVENTORY, ASSUMING 100 nCi/g TRU RADIONUCLIDES

Table A-3 lists the total simplified inventory (as of 1993) for the ALLW. The table is the result of adding the actinide entries from Table A-1 and the nonactinide entries from Table A-2. All of the waste is assumed to be in posttreatment waste packages with TRU radionuclide activity of 100 nCi/g, the upper limit for ALLW.

Table A-2. Development of simplified inventory of nonactinides.

Radionuclide	Inventory in TRU waste and ALLW in 1993 (Ci)	0.5 times inventory (ALLW only) (Ci)	Screened inventory (Ci)	Same activity after adjustment (Ci)
Ba-137m	2,246	1,123	1,123	1,125
C-14	<E-04	<E-04	—	—
Ce-144	27.06	13	—	—
Co-58	<E-04	<E-04	—	—
Co-60	94.3	47	47	47
Cr-51	<E-04	<E-04	—	—
Cs-134	110.7	55	—	—
Cs-137	2,003	1,001	1,001	1,125
Eu-150	<E-04	<E-04	—	—
Eu-152	0.068	0.034	—	—
Eu-154	0.71	0.35	—	—
Eu-155	0.19	0.1	—	—
Fe-55	1.132	0.57	—	—
Fe-59	<E-04	<E-04	—	—
H-3	0.8	0.4	—	—
Kr-85	6.857	3.4	—	—
MAP	1.28	0.64	—	—
MFP	483.3	242	242	—
Mn-54	0.23	0.12	—	—
Nb-95	<E-04	<E-04	—	—
Ni-63	3.572	1.79	—	—
Pm-147	27.26	14	—	—
Pr-144	27.18	14	—	—
Sr-90	1,778	889	889	1,013
Ta-182	<E-04	<E-04	—	—
Tc-99	0.0018	0.0009	—	—
Unid. β - γ	12.44	6	6	—
Y-90	2,019	1,010	1,010	1,013
Zn-65	<E-04	<E-04	—	—
Zr-95	<E-04	<E-04	—	—
Total	8,843	4,421	4,318	4,323

Note: Totals may not match exactly because of rounding.

Table A-3. Total simplified inventory, assuming 100 nCi/g TRU radionuclides.

Radionuclide	Inventory in 1993 (Ci)	TRU radionuclide inventory (Ci)
Ba-137m	1,125	—
Co-60	47	—
Cs-137	1,125	—
Sr-90	1,013	—
Y-90	1,013	—
Am-241	1,340	1,340
Cm-244	8	—
Pu-238	906	906
Pu-239	598	598
Pu-240	147	147
Pu-241	2,401	—
U-233	13	—
U-238	<u>0.0017</u>	<u>—</u>
Total	9,736	2,990

Note: Totals may not match exactly because of rounding.

A-7. TOTAL INVENTORY, ASSUMING 10 nCi/g OR 1,000 nCi/g TRU RADIONUCLIDES

The preceding analysis yields an upper-limit inventory, the situation in which every container has 100 nCi/g of TRU radionuclides. This section scales those inventories to the following situations: (a) every container has 10 nCi/g of TRU radionuclides, a lower-limit inventory for the ALLW; (b) every container has 1,000 nCi/g of TRU radionuclides, a situation in which the waste is actually TRU waste, not ALLW.

For the 10 nCi/g situation, it was assumed that the activity of *all* the actinides was reduced by a factor of 10. The nonactinide inventory was assumed not to change because of the manner in which the inventory was derived.

The results appear in Table A-4.

For the 1,000 nCi/g situation, it was assumed that the activity of *all* the actinides was increased by a factor of 10. The nonactinide inventory again was assumed not to change because of the manner in which the inventory was derived.

The results appear in Table A-5.

A-8. EFFECTS OF TREATMENT ON INVENTORY FOR VARIOUS FWFS

A-8.1 Results

As stated in Section A-4.4, the midpoint waste loading, which happened to match that of the cement FWFS, was used to develop a baseline actinide inventory. That inventory was established at 100 nCi of TRU radionuclides/g as an upper limit for ALLW. This section develops the corresponding radionuclide inventory for all of the FWFS.

Table A-6 presents the values of the parameters used to develop the inventory for each FWF. The parameters of waste loading, mass change associated with treatment, posttreatment density and volume, and the total inventory of radionuclides are indicated. The first column of Table A-6 also indicates in parentheses how each derived parameter was calculated.

Because the ranges of densities for the cements and for the glasses were relatively small, the respective average densities were used for these two waste categories. The density values are 1.95 g/cm³ for the cements and 2.65 g/cm³ for the glasses. This simplification was not used for the ceramics because the range of densities was larger.

As Table A-6 indicates, the same radionuclide inventory, in Ci, was used for all FWFS. Because of the different posttreatment masses of the FWFS, the TRU radionuclide concentration by mass must vary from one FWF to another. Only for cement is the value 100 nCi/g. The values for the other FWFS are either somewhat above or somewhat below 100 nCi/g. Section A-8.2 discusses this situation and its implications.

Table A-4. Total simplified inventory, assuming 10 nCi/g TRU radionuclides.

Radionuclide	Inventory in 1993 (Ci)	TRU radionuclide inventory (Ci)
Ba-137m	1,125	—
Co-60	47	—
Cs-137	1,125	—
Sr-90	1,013	—
Y-90	1,013	—
Am-241	134	134
Cm-244	0.8	—
Pu-238	91	91
Pu-239	60	60
Pu-240	15	15
Pu-241	240	—
U-233	1.3	—
U-238	0.00017	—
Total	4,865	300

Note: Totals may not match exactly because of rounding.

Table A-5. Total simplified inventory, assuming 1,000 nCi/g TRU radionuclides.

Radionuclide	Inventory in 1993 (Ci)	TRU radionuclide inventory (Ci)
Ba-137m	1,125	—
Co-60	47	—
Cs-137	1,125	—
Sr-90	1,013	—
Y-90	1,013	—
Am-241	13,400	13,400
Cm-244	80	—
Pu-238	9,060	9,060
Pu-239	5,980	5,980
Pu-240	1,470	1,470
Pu-241	24,010	—
U-233	130	—
U-238	0.017	—
Total	58,453	29,910

Note: Totals may not match exactly because of rounding.

Table A-6. Values of parameters used to develop inventory of ALLW in various final waste forms.

Parameter	More favorable and less favorable cement (FUETAP and PCC)	More favorable glass (ASG ^a)	Less favorable glass (BSG ^b)	More favorable ceramic (IEB4 ^c)	Less favorable ceramic (IEB ^d)
1. Waste loading, % by mass	40	60	30	60	60
2. Mass decrease factor due to incineration, etc.	0.75	0.75	0.75	0.75	0.75
3. Mass increase factor due to immobilization (100/row 1)	2.5	1.67	3.33	1.67	1.67
4. Mass change factor (row 2 × row 3)	1.88	1.25	2.5	1.25	1.25
5. Posttreatment mass, g (1.59E+10 g pretreatment × row 4)	2.99E+10	1.99E+10	3.98E+10	1.99E+10	1.99E+10
6. Posttreatment density, g/cm ³	1.95	2.65	2.65	3.5	3.0
7. Posttreatment volume, m ³ (row 5 × E-06/row 6)	1.53E+04	7.51E+03	1.50E+04	5.69E+03	6.63E+03
8. Radionuclide inventory, Ci	2,990 TRU 5,413 total actinides 4,323 nonactinides 9,736 total	Same	Same	Same	Same
9. TRU radionuclide conc. by mass, nCi/g (100 × 2.99E+10/row 5)	100	150	75	150	150

a. ASG—aluminosilicate glass.

b. BSG—borosilicate glass.

c. IEB—iron-enriched basalt.

d. IEB4—enhanced iron-enriched basalt.

If the simplified inventory entries in Table A-3 are divided by the total volume for each FWF in row 7 of Table A-6, the volumetric concentration of each radionuclide in each FWF is obtained. The results are given in Table A-7. These results constitute the inventory inputs to the release scenario calculations for each FWF.

A-8.2 An "Overspecified" System

In parametric studies, a complex set of parameters and their values can inadvertently become "overspecified." Numerous parameters may be specified independently, whereas in reality, certain relationships constrain the values among the parameters. To generate comparable results from one parametric run to another, inconsistencies of lesser importance may be accepted so that more important parameters have consistent values. Such is the case with the radionuclide inventories and the radionuclide concentrations by mass and volume for the various FWFs.

Let us assume that parametric computer runs are being conducted for different FWFs. If the radionuclide inventory is fixed and if the FWFs have different posttreatment masses, then the concentrations of TRU radionuclides by mass in the various FWFs must differ. The equation "radioactivity per unit mass = radioactivity/mass" relates these three parameters. If two of the parameters are specified, the third is automatically determined and cannot be picked arbitrarily.

Thus, total matchup of the parametric computer runs for different FWFs is not possible. It was considered realistic and essential that the total radionuclide inventory, in curies, be constant across all FWFs. Because the masses of the FWFs differ, the nCi/g of TRU radionuclides varies among the FWFs.

As discussed in Section A-4.4, a midpoint value of the waste loading was selected. The inventory was established so, for the FWFs that happened to have that waste loading, a value of 100 nCi/g of TRU radionuclides resulted. This is the conservative, upper-limit situation for the ALLW. The other FWFs have waste loadings somewhat higher or lower than that of the midpoint value, and posttreatment masses that vary correspondingly. Accordingly, the concentration of TRU radionuclides is somewhat higher or lower than 100 nCi/g for these FWFs. There is no way to match all of the values across the FWFs because the FWFs differ in waste loading, posttreatment mass, density, etc.

Both the intrusion scenarios and the groundwater scenario depend on the total radioactivity per unit volume of waste. The total radioactivity is identical for all FWFs. The FWF volumes differ, so the values of Ci/m³ differ, but the comparisons are on an equal footing.

In reality, during the treatment campaign, the mass differences of the various FWFs will result in some containers of treated waste crossing the boundaries representing 10 nCi/g and 100 nCi/g. In this sense, the parametric computer runs do not perfectly represent the realities of operational decisions that may change the classification of a given container from TRU waste to ALLW or from ALLW to LLW, for example. Nevertheless, the current runs are sufficiently consistent among the FWFs to provide results that are reasonably comparable.

Table A-7. Volumetric concentrations of each radionuclide in each final waste form, in Ci/m³, assuming 100 nCi/g TRU radionuclides for cement.

Radionuclide	More favorable and less favorable cement (FUETAP and PCC)	More favorable glass (ASG)	Less favorable glass (BSG)	More favorable ceramic (IEB4)	Less favorable ceramic (IEB)
Ba-137m	7.34E-02	1.50E-01	7.49E-02	1.98E-01	1.70E-01
Co-60	3.07E-03	6.26E-03	3.13E-03	8.27E-03	7.09E-03
Cs-137	7.34E-02	1.50E-01	7.49E-02	1.98E-01	1.70E-01
Sr-90	6.61E-02	1.35E-01	6.74E-02	1.78E-01	1.53E-01
Y-90	6.61E-02	1.35E-01	6.74E-02	1.78E-01	1.53E-01
Am-241	8.74E-02	1.78E-01	8.92E-02	2.36E-01	2.02E-01
Cm-244	5.22E-04	1.07E-03	5.33E-04	1.41E-03	1.21E-03
Pu-238	5.91E-02	1.21E-01	6.03E-02	1.59E-01	1.37E-01
Pu-239	3.90E-02	7.96E-02	3.98E-02	1.05E-01	9.02E-02
Pu-240	9.59E-03	1.96E-02	9.79E-03	2.59E-02	2.22E-02
Pu-241	1.57E-01	3.20E-01	1.60E-01	4.22E-01	3.62E-01
U-233	8.48E-04	1.73E-03	8.66E-04	2.29E-03	1.96E-03
U-238	1.11E-07	2.26E-07	1.13E-07	2.99E-07	2.56E-07
Total	6.35E-01	1.30E+00	6.48E-01	1.71E+00	1.47E+00

A-9. REFINED INVENTORY FOR NONACTINIDES

A-9.1 Introduction

Section A-5 of this appendix described how the nonactinide inventory that was used in almost all of the PA calculations was developed. Runs performed during the first phase of the study indicated a considerable dependence of some intrusion doses on the nonactinide inventory. There was a large uncertainty in that nonactinide inventory.

This section describes an improved set of assumptions used to develop a refined nonactinide inventory that is more realistic. The latter approach was used for a few PA calculations to evaluate the effect of these particular inventory assumptions. The inventory of actinides was not altered from that in Table A-3.

The distinction between the two methods lies in whether the RH ALLW and CH ALLW are addressed jointly or separately. The approach in Section A-5 was basically to (a) assume that,

before waste treatment, all nonactinides are in the RH waste, (b) note that about 50% of the RH waste volume is ALLW, (c) assume that 50% of the RH activity is therefore in the RH ALLW, and (d) spread the resulting activity uniformly over all posttreatment CH and RH ALLW. The simplified approach was useful for scoping evaluations. However, it tends to overestimate the concentration of nonactinides in the CH ALLW and underestimate the concentration of nonactinides in the RH ALLW. In the following refined analysis, the RH ALLW was assumed to be treated separately from the CH ALLW, and the nonactinide inventories were estimated separately.

It was assumed again that the distribution of nonactinides is independent of the distribution of TRU radionuclides. Thus, if a given fraction by volume of the CH waste is CH ALLW, then the same fraction of the nonactinides in the CH waste is in the CH ALLW. The same assumption was made for the RH waste, although the fraction differs.

Separate RWMIS printouts (covering combined ALLW and TRU waste) were produced for the CH waste and for the RH waste.

A-9.2 CH Waste

The process for developing the CH inventory is depicted in Table A-8. The first two columns list the nonactinide inventory in the CH waste at the time of emplacement according to RWMIS.

The next column gives an approximate value for the activity decayed to 1993. The values are approximate because, for the nonactinides, the ratios of decayed activity to emplaced activity derived in Atwood and Schlafman (1993) for the total stored waste inventory were assumed here to apply also to the CH and RH inventories individually. Basically, this simplifying assumption implies that the CH and RH were emplaced at the same rate, in terms of activity, from 1970 through 1993. The assumption was used in lieu of performing the detailed decay calculations for every year of waste emplacement, as was done by Atwood and Schlafman.

It was next assumed that, since the CH ALLW has approximately 38% of the total CH volume,^a it also has 38% of the total CH activity of nonactinides. Thus, the entries in the fourth column were produced by multiplying those in the third column by 0.38.

The inventory was subjected to screening as in Section A-5 of this appendix to reduce the number of radionuclides to only those of potential importance. All radionuclides with <1 Ci of activity were deleted. C-14, Ce-144, Co-58, Cr-51, Cs-134, Eu-150, Eu-152, Eu-154, Eu-155, Fe-55, H-3, Mn-54, Ni-63, Tc-99, Zn-65, and Zr-95 were thereby eliminated. The last screening step was to delete the 8.78 Ci of Pm-147, based on application of the screening method of Seitz

a. The volume estimates are as follows, in m³: RH total= 76 (from RWMIS); RH ALLW = 31 (from RWMIS); CH total= 63,729 (from Raivo et al. 1993) - 76 = 63,653; CH ALLW = 24,413 (from Raivo et al. 1993) - 31 = 24,382. Thus, the fraction of CH waste that is ALLW is 24,382/63,653 = 0.38. The fraction of RH waste that is ALLW is 31/76 = 0.41.

Table A-8. Development of refined inventory of nonactinides for CH ALLW.

Radionuclide	Inventory in CH TRU waste and CH ALLW at time of emplacement (Ci)	Inventory in 1993 (Ci)	0.38 times inventory (ALLW only) (Ci)	Screened inventory (Ci)	Same activity after adjustment (Ci)
Ba-137m	0	61.75	23.5	23.5	54.3
C-14	<E-04	<E-04	<E-04	—	—
Ce-144	9.021	0.06	0.02	—	—
Co-58	0.83	<E-04	<E-04	—	—
Co-60	192	77.8	29.6	29.6	29.6
Cr-51	1.45	<E-04	<E-04	—	—
Cs-134	0.002	0.0002	<E-04	—	—
Cs-137	72.94	61.75	23.5	23.5	54.3
Eu-150	<E-04	<E-04	<E-04	—	—
Eu-152	0.17	0.067	0.025	—	—
Eu-154	0.0002	<E-04	0.17	—	—
Eu-155	<E-04	<E-04	<E-04	—	—
Fe-55	0.0006	0.0002	<E-04	—	—
H-3	1.95	0.80	0.30	—	—
MAP	1.75	0.39	0.15	0.15	—
MFP	195.4	149.9	57.0	57.0	—
Mn-54	0.14	0.0003	0.0001	—	—
Ni-63	0.0001	0.0001	<E-04	—	—
Pm-147	464	23.1	8.78	—	—
Sr-90	0.3	0.25	0.095	0.095	30.9
Tc-99	0.0018	0.0018	0.0007	—	—
Unid. β - γ	14.89	12.44	4.7	4.7	—
Y-90	0	0.25	0.095	0.095	30.9
Zn-65	0.001	<E-04	<E-04	—	—
Zr-95	<u>66.8</u>	<u><E-04</u>	<u><E-04</u>	<u>—</u>	<u>—</u>
Total	1,021	389	148	139	200

and Kocher (1993) for the groundwater pathway and on computer runs for the intrusion scenarios. The screened inventory appears in the fifth column of Table A-8. The total activity has been reduced from 148 Ci to 139 Ci, primarily due to screening out the 8.78 Ci of Pm-147.

Two final steps were involved in developing the refined nonactinide inventory for the CH ALLW. One step was to apportion the MFP and unidentified β - γ entries, both of which were taken to be 50% Sr-90 and 50% Cs-137, and add the resulting activity to the respective radionuclides indicated. The extremely small activity (0.15 Ci) of mixed activation products (MAP) was neglected compared to the much larger activity of Co-60, its most likely predominant constituent. The other step was to readjust the Ba-137m activity to match that of Cs-137, and the Y-90 activity to match that of Sr-90, based on decay equilibrium considerations. The result is in the last column of Table A-8.

Comparison of the last column of Table A-8 (the refined inventory for nonactinides) with the last column of Table A-2 (the simplified inventory for nonactinides) shows a large reduction in the nonactinide inventory in the CH ALLW. The Cs-137 activity is reduced to about 5% of its previous value; Sr-90, to about 3%. (The Co-60 activity is still about 63% of its earlier value, however.) The list of radionuclides that survived the screening is unchanged.

The volume of the CH ALLW is virtually identical to that of the total ALLW (24,382 m³ versus 24,413 m³). Therefore, the concentrations of the nonactinides obtained in the refined inventory are reduced in proportion to their respective reductions in radioactivity.

A-9.3 RH Waste

The process for developing the RH waste inventory is similar to that for the CH waste and is depicted in Table A-9. The first two columns list the nonactinide inventory in the RH waste at the time of emplacement, according to RWMIS. The next column gives an approximate value for the activity decayed to 1993, using the approach described for the CH waste in Section A-9.2 of this appendix.

It was next assumed that, since the RH ALLW has approximately 41% of the total RH volume (31 of 76 m³, based on the RWMIS information), it also has 41% of the total RH activity of nonactinides. Thus, the entries in the fourth column were produced by multiplying those in the third column by 0.41. (This fraction differs somewhat from the 0.5 value used for Table A-2 because it is based solely on the RWMIS value, which is probably the most reliable among the estimates that have been made.)

The inventory was subjected to screening (as in Section A-5 of this appendix) to reduce the number of radionuclides to only those of importance. All radionuclides with <1 Ci of activity were deleted. Co-58, Cr-51, Eu-154, Eu-155, Fe-55, Fe-59, Mn-54, Nb-95, Rh-106, Ru-106, Sb-125, Ta-182, and Zr-95 were thereby eliminated. The last screening step was to delete the Ce-144, Cs-134, Kr-85, Ni-63, and Pm-147 based on application of the screening method of Seitz and Kocher (1993) for the groundwater pathway and on computer runs for the intrusion scenarios. The screened inventory appears in the fifth column of Table A-9 and totals 98% of that before screening.

Table A-9. Development of refined inventory of nonactinides for RH ALLW.

Radionuclide	Inventory in RH TRU waste and RH ALLW at time of emplacement (Ci)	Inventory in 1993 (Ci)	0.41 times inventory (ALLW only) (Ci)	Screened inventory (Ci)	Same activity after adjustment (Ci)
Ba-137m	97.33	1,984	813	813	882
Ce-144	3,890	27	11.1	—	—
Co-58	112.4	<E-04	<E-04	—	—
Co-60	40.64	16.5	6.8	6.8	6.8
Cr-51	7.01	<E-04	<E-04	—	—
Cs-134	929.6	110.7	45.4	—	—
Cs-137	2,343	1,984	813	813	882
Eu-154	1.555	0.71	0.29	—	—
Eu-155	0.78	0.19	0.08	—	—
Fe-55	3.858	1.13	0.46	—	—
Fe-59	5.34	<E-04	<E-04	—	—
Kr-85	9.362	6.86	2.81	—	—
MAP	4.0	0.89	0.36	—	—
MFP	434.5	333	137	137	—
Mn-54	115.4	0.23	0.09	—	—
Nb-95	0.72	<E-04	<E-04	—	—
Ni-63	3.694	3.572	1.46	—	—
Pm-147	84.13	4.18	1.71	—	—
Rh-106	1.3	<E-04	<E-04	—	—
Ru-106	105.4	0.19	0.08	—	—
Sb-125	6.087	1.647	0.68	—	—
Sr-90	2,115	1,777	729	729	798
Ta-182	0.735	<E-04	<E-04	—	—
Y-90	101.8	1,777	729	729	798
Zr-95	<u>0.432</u>	<u><E-04</u>	<u><E-04</u>	<u>—</u>	<u>—</u>
Total	10,414	8,029	3,292	3,228	3,367

Two final steps were involved in developing the refined nonactinide inventory for the RH ALLW. One step was to apportion the MFP, which were taken to be 50% Sr-90 and 50% Cs-137, and add the resulting activity to the respective radionuclides indicated. The extremely small activity (0.36 Ci) of mixed activation products (MAP) was neglected compared to the much larger activity of Co-60, its most likely predominant constituent. The other step was to readjust the Ba-137m activity to match that of Cs-137, and the Y-90 activity to match that of Sr-90, based on decay equilibrium considerations. The result is in the last column of Table A-9.

The activity values in the last column of Table A-9 are slightly smaller than the corresponding values in the last column of Table A-2. (One exception is Co-60, which is much smaller—only about 14% of its earlier value.) This result was expected because (a) the activity in the CH ALLW was addressed separately from the RH ALLW and (b) the volume (and activity) of the RH ALLW was assumed to be only 41%, rather than the 50% used previously, of the total for the RH waste.

The volume of the RH ALLW is only about 0.13% of the total ALLW (31 m³ versus 24,413 m³). Therefore, the concentrations of the nonactinides obtained using the refined assumptions are about $1/0.0013 = 770$ times larger than the values developed using the simplified methods described in Section A-5 of this appendix.

Tables A-8 and A-9 indicate that approximately 95% of the nonactinide activity in the TRU waste and ALLW in 1993 is in the RH waste: $8,029/(8,029 + 389)$. The simplified assumption discussed in Section A-5.1 was 100%.

A-9.4 Conclusions

The refined assumptions result in the activity and the activity concentration of the nonactinides generally decreasing by factors of 20 to 30 (depending on the radionuclide) for the CH ALLW. This decrease would be expected to reduce the receptor dose substantially for the intruder scenarios.

For the RH ALLW, the refined assumptions result in little change in the nonactinide activity inventory except an increase by a factor of about 770 in the activity concentration. This increase would be expected to substantially increase the receptor dose for intruder scenarios in which penetration of the waste occurs.

A-10. ADDITIONAL REFINEMENTS TO NONACTINIDE INVENTORY

A-10.1 Introduction

This section reports two investigations undertaken during the third phase of the study to produce "additional refinements" to the nonactinide inventory:

- In one investigation, the RWMIS entries for each untreated waste shipment to the TSA containing nonactinides totaling >1 Ci were examined to determine whether the

containers in the shipment would be expected to be TRU waste or ALLW, and CH waste or RH waste. This was done rather than making broad assumptions about the distribution of nonactinides in TRU waste versus ALLW, and in CH waste versus RH waste, as had been done in the two preceding inventories. Based on these new determinations, the estimate of the ALLW nonactinide inventory was revised.

- In the other investigation, additional information on the nonactinide inventory was obtained from sources other than RWMIS.

The resulting "inventory with additional refinements" is described in this section. It was used in a limited number of runs that are so identified. The actinide inventory was not altered from that in Table A-3.

A-10.2 Distribution of Nonactinides Between ALLW and TRU Waste Based on the Examination of the Records for Individual Shipments

The RWMIS data on individual waste shipments were examined to identify the nonactinides that were believed to be in the ALLW, as opposed to the TRU waste. This search, which was performed at the level of waste shipments rather than at the level of individual waste containers, was performed as follows.

A search of the RWMIS database included all shipments of waste placed into storage at the TSA. Separate searches were conducted for CH waste, which was placed on the TSA pads, and RH waste, which was placed in the ILTSF.

All shipments with a total nonactinide inventory of >1 Ci were flagged. Shipments with this flag were then sorted into those expected to be assayed as TRU waste versus those expected to be ALLW. The sorting was done by calculating (in accordance with DOE Order 5820.2A) the total TRU activity in the shipment and dividing by the mass of the waste in the shipment. To be conservative, any shipments with an average activity of <200 nCi TRU/g of waste were included in the ALLW. Using 200 nCi/g rather than the prescribed cutoff of 100 nCi/g provided a conservatism of a factor of 2 in estimating which shipments are ALLW rather than TRU waste.

The number of CH waste shipments identified under the above criteria was 53 from five generators. For RH waste, the number of such shipments was 63 from three generators.

All individual nonactinide entries >0.1 Ci were then summed over all such shipments. The results are given in Tables A-10 and A-11 for CH waste and RH waste, respectively. As in Section A-5, entries listed as MFP or unidentified β - γ were assumed to be 50% Cs-137 and 50% Sr-90. Also as in Section A-5, the activities of Ba-137m and Y-90 were adjusted to match those of Cs-137 and Sr-90, respectively.

Table A-10. Nonactinide inventory assigned to CH ALLW by use of RWMIS records of individual shipments.

Radionuclide	Reported activity (Ci)	Activity after adjustment (Ci)
Ba-137m	—	88.6
Ce-144	1.0	1.0
Co-60	39.2	39.2
Cs-137	16.5	88.6
MFP	137.8	—
Sr-90	—	72.1
Unid. β - γ	6.3	—
Y-90	—	72.1
Zr-95	11.9	11.9
Total	212.7	373.5

Table A-11. Nonactinide inventory assigned to RH ALLW by use of RWMIS records of individual shipments.

Radionuclide	Reported activity (Ci)	Activity after adjustment (Ci)
Ba-137m	—	165.0
Ce-144	165.4	165.4
Co-58	13.2	13.2
Co-60	1.4	1.4
Cr-51	1.0	1.0
Cs-134	58.1	58.1
Cs-137	149.0	165.0
Fe-59	1.3	1.3
MFP	32.0	—
Mn-54	13.4	13.4
Nb-95	0.3	0.3
Sr-90	102.4	118.4
Y-90	—	118.4
Zr-95	0.2	0.2
Total	537.7	821.1

A-10.3 Additional Inventory Information from Sources other than RWMIS

The shipping records and their compilation as the RWMIS database often provide incomplete information on the radionuclides in the waste. Other sources of information sometimes contain useful data on the identity and activity of the radionuclides. Several potential sources of information on the nonactinides in the ALLW were investigated. That new information is discussed below.

A-10.3.1 Tritium

Based on RWMIS data, Atwood and Schlafman (1993) listed an activity of only 1.95 Ci of tritium (H-3) in the waste stored in the TSA. Radioactive decay had reduced the activity, recorded in 1977, to 0.7978 Ci by 1993. Examination of the RWMIS entry indicates that the waste containing the H-3 is likely to be ALLW, not TRU waste.

As described below, the present task led to the conclusion that additional H-3, not reported in RWMIS, is in the stored waste.

Two incidents at the Rocky Flats Plant resulted in tritium-contaminated waste being sent to the TSA (Clements 1984). Nearly all of the tritium in the stored waste is believed to exist in waste generated during the cleanup and decontamination work stemming from these two incidents.

Information was compiled in Clements (1984) on the tritium-contaminated waste from these incidents, based primarily on operator interviews. That document was used to develop the following inventory of tritium in the ALLW.

Tritium-bearing waste associated with the first incident was shipped to the TSA in late 1973 and early 1974. The great majority of the tritium is located in a total of 11 drums. Very small amounts of tritium (possibly up to the mCi range) were in drums shipped later.

Of the 11 drums, 9 were shown by assay at Rocky Flats Plant to have at least 1 g Pu and are almost certainly TRU waste, not ALLW. Two drums were shown by assay to have <1 g Pu. These two drums could conceivably be ALLW. (For a 300-lb drum, for example, about 0.17 g of weapons-grade Pu would represent a concentration of 100 nCi TRU/g.) Conservatively, it was assumed that these two drums were ALLW.

In the absence of more definitive information, the distribution of tritium throughout the 11 drums was assumed to be uniform. The activity of tritium in the two ALLW drums was thus estimated to be 2/11 of the total of approximately 600-800 Ci for the 11 drums, or 109-145 Ci. For the PA inventory, the conservative value assumed was 145 Ci as of late 1973.

The waste form in one of the two assumed ALLW drums resulting from the first incident is wet combustibles. In the other drum, the waste form is leaded rubber gloves. A conservative assumption indicated that the release mechanism for tritium was surface washoff. If the tritium

turns out to be a substantial contributor to dose in the PA, then a more realistic release mechanism, such as a dissolution process, may be used.

Tritium-bearing waste associated with the second incident is located in two drums shipped in 1981. These drums may or may not be ALLW. It was assumed here that both drums were ALLW. The amount of tritium in each of these drums was estimated to be 1 Ci, for a total of 2 Ci. The waste is in the form of dry combustibles, and a surface washoff release mechanism was assumed.

Based on the above information, the total activity of tritium in the ALLW "inventory with additional refinements" was assumed conservatively to be $145 + 2 + 1.95 = 149$ Ci at the time of generation. An amount of 145 Ci was generated in late 1973, plus 2 Ci in 1981, plus the 1.95 Ci listed in RWMIS that was recorded in 1977. A surface washoff release mechanism was assumed for the initial performance evaluations. The drums containing H-3 were assumed to be CH waste. No credit was taken for removal of H-3 from the waste during waste treatment.

A-10.3.2 Nonactinides in Radiation Sources

Clements and Darnell's study (1994), which is based on studies performed in 1979 and 1980, provides data that are not in RWMIS on radiation sources in RFP waste received at the RWMC. Separate listings are given for the larger and smaller sources. Some of these sources are in the stored waste, and some are in the buried waste. Of those in the stored waste, some may be in the ALLW and some in the TRU waste.

The data in Table A-12 on the nonactinides (plus Ra-226) in the radiation sources from RFP were obtained from Clements and Darnell (1994). All entries with an activity of 0.1 Ci or greater were included.

Waste packages shipped before or during October 1970 are generally considered to have been buried in the SDA rather than stored in the TSA. Therefore, the drums in the first, third, and fourth entries were assumed to have been buried and are outside the scope of this study.

The second entry is probably not TRU waste because it appears that only radiation sources, lead, and concrete were placed into this drum. Apparently, these particular sources were placed in sole-purpose packages, rather than being added to existing drums already containing waste.

The only nonactinide listed in RWMIS for stored waste from Rocky Flats Plant is the 1.95 Ci of H-3 discussed in A-10.3.1. It is associated with waste retrieved from Pits 11 and 12 and placed into storage. Based on Card (1977), that waste was buried in Pits 11 and 12 in 1970. Therefore, the 1.95 Ci of H-3 from RFP that is listed in RWMIS does not include any of the H-3 sources in the fifth through ninth entries listed above.

Thus, all of the entries except the first, third, and fourth are (a) in the stored waste, (b) not included in the RWMIS printouts used in the previous ALLW PA inventories, and (c) because of the years in which they were shipped, not included in the H-3 discussed in Section A-10.3.1.

Table A-12. Nonactinides plus Ra-226 in radiation sources received from the Rocky Flats Plant.

Shipping date	Package number	Nonactinide activity	Form	TRU waste?
6-18-70 ^a	Drum 444-679	174 Ci Co-60 214 Ci Cs-137	Lead shielded	Probably not
5-9-73	Drum 68-8	64 Ci Co-60	Lead shielded	Probably not
10-70 ^a	Drum (no number)	0.36 H-3	Unknown	Unknown
10-70 ^a	Drum (no number)	0.1 Ci Ra-226	Unknown	Unknown
1975	Box 1544-85251	5 Ci H-3	Unknown	Unknown
1975	Drum 36-01846	0.25 Ci H-3	Scandium tritide	Unknown
1977	Drum D05548	0.25 Ci H-3	Unknown	Unknown
3-31-77	Drum 742-25064	3.1 Ci H-3	Unknown	Unknown
8-78	Drum 742-25667	1.5 Ci H-3	Scandium tritide	Unknown

a. Assumed to be in buried waste; not included in this study.

Based on the preceding discussion, the "inventory with additional refinements" was supplemented to include the activity listed in Table A-13.

The 64 Ci of Co-60 were assumed to be in the ALLW. The containers totaling 10 Ci of H-3 were assumed to be in the ALLW. These activities do not reflect radioactive decay.

It was assumed that the H-3 is in CH waste. For the Co-60, it was assumed that during treatment, the source will be removed from its lead-shielded CH drum and addressed with the RH waste.

A-10.3.3 Possible Presence of Other Radionuclides

The radionuclides C-14, I-129, and Tc-99 are often present in small activities in waste generated as a result of the operation of nuclear reactors. The activities of these radionuclides are difficult to measure; they are often not reported or underreported in waste shipping records.

Table A-13. Nonactinides in radiation sources in stored ALLW received from Rocky Flats Plant.

Radionuclide	Activity	Release mechanism for untreated waste
Co-60	64 Ci	Entrapped in metallic matrix
H-3	10 Ci	Conservatively, surface washoff was assumed; if the H-3 impacts the dose, a more realistic, corrosion-based mechanism involving scandium tritide can be used

Activities of $4\text{E-}5$ Ci and $1.8\text{E-}3$ Ci for C-14 and Tc-99, respectively, were reported in RWMIS for the total of all waste (TRU waste plus ALLW) in the TSA (Atwood and Schlafman 1993). No I-129 was reported.

The radionuclides C-14, I-129, and Tc-99 are potentially of interest in PAs because they are very long-lived and are relatively mobile in groundwater. To project their potential effect on doses in this preliminary parametric PA, a conservative estimate of their possible activity in the ALLW was developed as follows.

The activities of these difficult-to-measure radionuclides are often estimated by comparing them with the activities of radionuclides that are readily measured. The activity of C-14 is compared with that of Co-60; the activities of I-129 and Tc-99 are compared with that of Cs-137. Typical values of these activity ratios, often called scaling factors, have been determined (EPRI 1987) by evaluating a large quantity of laboratory data. There is considerable scatter in the data for these three particular radionuclides.

The scaling factor approach has been used to attribute activities of these three radionuclides to some of the waste buried in the RWMC Subsurface Disposal Area (Lockheed Idaho 1995). The conservative estimates developed here were based on the use of scaling factors taken from that reference.

The scaling factor for C-14 in INEL buried waste typically ranges from $1\text{E-}3$ to $1\text{E-}2$ (Lockheed Idaho 1995), depending on the nature of the waste. If one conservatively multiplies the largest of these values, $1\text{E-}2$, by the estimated total activity of 104.6 Ci of Co-60 from Tables A-10, A-11, and A-13, the result is 1.0 Ci of C-14.

The scaling factor for Tc-99 in INEL buried waste typically ranges from $1\text{E-}5$ to $2\text{E-}4$ (Lockheed Idaho 1995), depending on the nature of the waste. If one conservatively multiplies the largest of these values, $2\text{E-}4$, by the estimated total activity of 253.6 Ci of Cs-137 from Tables A-10, A-11, and A-13, the result is 0.05 Ci of Tc-99.

The scaling factor for I-129 in INEL buried waste typically ranges from $3\text{E-}8$ to $7\text{E-}8$ (Lockheed Idaho 1995), depending on the nature of the waste. If one conservatively multiplies the largest of these values, $7\text{E-}8$, by the estimated total activity of 253.6 Ci of Cs-137 from Tables A-10, A-11, and A-13, the result is $1.8\text{E-}5$ Ci of I-129.

Despite their very small activities, C-14, Tc-99, and I-129 are potentially important contributors to the dose via the groundwater transport pathway. Therefore, the conservatively estimated activities of 1.0 Ci, 0.05 Ci, and $1.8\text{E-}5$ Ci, respectively, were retained for the groundwater pathway screening evaluation discussed in Section A-10.5. Because it is not known whether these three radionuclides could be in the CH waste or the RH waste, their activities were conservatively included in both types of waste but were counted only once in the total nonactinide inventory for the CH waste plus RH waste.

A-10.4 Nonactinide Inventory with Additional Refinements

This section presents the tables for the "inventory with additional refinements" and describes how they were compiled.

The nonactinide "inventory with additional refinements" is presented in Table A-14 for the CH portion, Table A-15 for the RH portion, and Table A-16 for the combined CH and RH contributions. These nonactinide inventories, combined with the unchanged actinide inventory from Table A-3, were used in a limited number of runs that are so identified.

The second column of each table lists the activity of each radionuclide at the time of generation. These entries were compiled from Tables A-10, A-11, and A-13, and the discussion in Section A-10.3.3.

A simplified approach was used to estimate the radioactive decay of the nonactinides from the time of generation through 1993. The approach is indicated in the third and fourth columns of Tables A-14 and A-15. Nonactinides were identified in RWMIS in a total of 116 shipments of ALLW over a period of many years. Half of the radioactivity in the CH waste in the TSA had been received by about 1980; half of the radioactivity in the RH waste had been received by about 1985 (DOE 1994). As an approximation, the nonactinides in the CH waste were understood to have been generated in 1980, and the nonactinides in the RH waste were understood to have been generated in 1985.

Exceptions to the above simplification involved a small number of additional ALLW shipments that, as discussed in Section A-10.3, contained H-3 or radiation sources and were not reported in RWMIS. For the shipments containing major entries of H-3 or radiation sources, the exact years of generation were used herein to calculate radioactive decay. Approximations were used for the dates of generation of shipments containing the smaller activities.

The next column of Tables A-14 through A-16 shows the activity after screening out certain radionuclides from the inventory. If a radionuclide would make a negligibly small contribution to the groundwater transport dose and to the intrusion scenario doses, then the radionuclide was deleted from the inventory. The method for making this determination was as follows. Section 2.5 presents the screening evaluation for the dose from groundwater transport. In those evaluations, all nonactinides in the inventory with additional refinements were screened out except H-3, C-14, Tc-99, and I-129. These four radionuclides were retained in the inventory because of their potential contribution to the groundwater transport dose. Other radionuclides with activities <1 Ci in 1993 were deleted from the inventory because they were considered (a) not to be substantial contributors to the intrusion scenario dose and (b) not to be substantial contributors to the groundwater transport dose. This led to deleting Ce-144, Co-58, Cr-51, Fe-59, Mn-54, Nb-95, and Zr-95 from the inventory. All other indicated nonactinides were retained for the intrusion evaluations. H-3, C-14, Tc-99, and I-129 were the only nonactinides included in the groundwater transport evaluations. The entries in this column of the table were used in the PA runs.

Table A-14. Nonactinide inventory with additional refinements for CH ALLW.

Radionuclide	Activity at time of generation (Ci)	Approximate year of generation	Activity in 1993 (Ci)	Activity after screening (Ci)	Activity in refined inventory ^a as of 1993 (Ci)
Ba-137m	88.6	1980	67.3	67.3	54.3
C-14	<1.0	1980	<1.0	1.0	—
Ce-144	1.0	1980	2.3E-5	—	—
Co-60	39.2	1980	8.1	8.1	29.6
Cs-137	88.6	1980	67.3	67.3	54.3
H-3	159	145 Ci in 1973 5 Ci in 1975 7 Ci in 1977 2 Ci in 1981	55.8	55.8	—
I-129	<1.8E-5	1980	<1.8E-5	1.8E-5	—
Sr-90	72.1	1980	54.1	54.1	30.9
Tc-99	<0.05	1980	<0.05	0.05	—
Y-90	72.1	1980	54.1	54.1	30.9
Zr-95	11.9	1980	≈0	—	—
Total	533.5	NA	307.7	307.7	200

a. Taken from Table A-8.

Table A-15. Nonactinide inventory with additional refinements for RH ALLW.

Radionuclide	Activity at time of generation (Ci)	Approximate year of generation	Activity in 1993 (Ci)	Activity after screening (Ci)	Activity in refined inventory ^a as of 1993 (Ci)
Ba-137m	165.0	1985	140.5	140.5	882
C-14	<1.0	1985	<1.0	1.0	—
Ce-144	165.4	1985	0.325	—	—
Co-58	13.2	1985	≈0	—	—
Co-60	65.4	64 Ci in 1973 1.4 Ci in 1985	5.82	5.82	6.8
Cr-51	1.0	1985	≈0	—	—
Cs-134	58.1	1985	5.5	5.5	—
Cs-137	165.0	1985	140.5	140.5	882
Fe-59	1.3	1985	≈0	—	—
I-129	<1.8E-5	1985	<1.8E-5	1.8E-5	—
Mn-54	13.4	1985	0.046	—	—
Nb-95	0.3	1985	≈0	—	—
Sr-90	118.4	1985	100.2	100.2	798
Tc-99	<0.05	1985	<0.05	0.05	—
Y-90	118.4	1985	100.2	100.2	798
Zr-95	0.2	1985	≈0	—	—
Total	886.1	NA	494.1	493.8	3366.8

a. Taken from Table A-9.

Table A-16. Nonactinide inventory with additional refinements for CH plus RH ALLW.

Radionuclide	Activity at time of generation (Ci)	Activity in 1993 (Ci)	Activity after screening (Ci)	Activity in refined inventory ^a as of 1993 (Ci)	Activity in simplified inventory ^b as of 1993 (Ci)
Ba-137m	253.6	207.8	207.8	936.3	1125
C-14	<1.0	<1.0	1.0	—	—
Ce-144	166.4	0.325	—	—	—
Co-58	13.2	≈0	—	—	—
Co-60	104.6	13.92	13.92	36.4	47
Cr-51	1.0	≈0	—	—	—
Cs-134	58.1	5.5	5.5	—	—
Cs-137	253.6	207.8	207.8	936.3	1125
Fe-59	1.3	≈0	—	—	—
H-3	159	55.8	55.8	—	—
I-129	<1.8E-5	<1.8E-5	1.8E-5	—	—
Mn-54	13.4	0.046	—	—	—
Nb-95	0.3	≈0	—	—	—
Sr-90	190.5	154.3	154.3	828.9	1013
Tc-99	<0.05	<0.05	0.05	—	—
Y-90	190.5	154.3	154.3	828.9	1013
Zr-95	12.1	≈0	—	—	—
Total	1418.6	800.8	800.5	3566.8	4323

a. Taken from Tables A-8 and A-9.

b. Taken form Table A-3.

A-10.5 Comparisons of the Three Successive Nonactinide Inventories

The CH and RH portions of the nonactinide "refined inventory" from Tables A-8 and A-9 are also given in Tables A-14 and A-15, respectively. The combined CH and RH portions of the refined inventory (from Table A-8 plus Table A-9) are given in Table A-16. This allows a comparison of the "refined inventory" and the "inventory with additional refinements."

For completeness, the nonactinide "simplified inventory," taken from Table A-3, is also given in Table A-16.

Table A-16 shows that the trend in the estimate of the total nonactinide inventory, from the simplified inventory to the refined inventory to the inventory with additional refinements, is downward. The total activity was reduced in the second inventory by 17% and in the third inventory by a total of 81%. The major reductions were in the predominant radionuclides Cs-137 and Ba-137m, Sr-90 and Y-90, and Co-60. The inventory with additional refinements added five radionuclides not addressed in the two previous inventories: H-3, C-14, Tc-99, and I-129, plus Cs-134.

Tables A-14 and A-15 show that, although the nonactinide activity in the CH waste increased by about 50% (from 200 Ci to 307.7 Ci) in the inventory with additional refinements, the activity in the RH waste decreased by a factor of almost 7 (from 3366.8 Ci to 493.8 Ci). The other substantial difference is in the presence of small activities of C-14, Tc-99, and I-129 (and H-3 for the CH waste) in the inventory with additional refinements. The activities of these four radionuclides were derived from information sources other than RWMIS.

REFERENCES FOR APPENDIX A

- Atwood C. L., and M. J. Schlafman, 1993, *Empirical Distributions of Radionuclides from RWMIS Data*, EG&G Idaho, Inc., EGG-RAAM-10741, April.
- Card, D. H., 1977, *History of Buried Transuranic Waste at INEL*, EG&G Idaho, Inc., WMP 77-3, March.
- Clements, T. L., Jr., 1984, "Tritium-Contaminated Stored TRU Waste," letter to K. B. McKinley, EG&G Idaho, Inc., TLC-6-84, May 14.
- Clements, T. L., Jr., and G. R. Darnell, 1994, "Rocky Flats Plant Characterization (1954-1980)," EG&G Idaho, Inc., Engineering Design File RWMC-761, July.
- DOE, 1994, *Integrated Data Base for 1993: U.S. Spent Fuel and Radioactive Waste Inventories, Projections, and Characteristics*, U.S. Department of Energy, DOE/RW-0006, Rev. 9, March.
- EPRI, 1987, *Updated Scaling Factors in Low Level Radwaste*, EPRI NP-5077, Impell Corporation, March.
- Lockheed Idaho, 1995, *A Comprehensive Inventory of Radiological and Nonradiological Contaminants in Waste Buried in the Subsurface Disposal Area of the INEL RWMC During the Years 1984-2003*, Lockheed Idaho Technologies Company, INEL-95/0135, May, 3 volumes.
- Raivo, B. D., T. L. Eddy, and G. L. Anderson, 1993, *Statement of Work: Commercial Transport and Treatment of Alpha-Contaminated Mixed Low-Level Radioactive Waste (draft)*, Attachment 1 to *Solid Alpha-Contaminated Low-Level Radioactive Waste Transport/Treatment Services Request for Proposal*, SP-SOW-01, June.
- Seitz, R. R., and Kocher, D. C., 1993, "On Establishing Waste Concentration Limits for Low-level Radioactive Waste Disposal," *Proceedings of the Symposium on Waste Management, Tucson, Arizona, March*.

Appendix B

Refined Model of Untreated Alpha-Contaminated Low-Level Waste Stored at the INEL

Appendix B

Refined Model of Untreated Alpha-Contaminated Low-Level Waste Stored at the INEL

B-1. BACKGROUND

After treatment, alpha-contaminated low-level waste (ALLW) at the Idaho National Engineering Laboratory (INEL) is expected to be fairly homogeneous, depending on the operational practices at the treatment facility. By contrast, the untreated ALLW is highly heterogeneous in its physical, chemical, and radiological characteristics.

No waste description of manageable complexity for this study can describe the ALLW precisely. Therefore, an approximate model representing the radiological inventory and release characteristics of the waste was developed. Grouping the untreated waste into categories with similar characteristics was necessary, together with assuming homogeneity of the waste within a category. This appendix describes the method used.

The refined radionuclide inventory used for the untreated ALLW coincides with that developed in Appendix A, Table A-3 for the actinides and Tables A-8 and A-9 (the refined inventory) for the nonactinides. Thus, the discussion in this appendix does not involve a new inventory but a regrouping of the same refined inventory discussed in Appendix A.

B-2. OPTIONS FOR MODELING THE WASTE COMPOSITION

Several options exist for modeling the composition of the ALLW. On one extreme, the waste could be modeled as one homogeneous mass. As stated above, the untreated ALLW is so heterogeneous that such an approach is obviously unacceptable.

At the other extreme, the untreated ALLW could be modeled, in theory, on a container-by-container basis. Homogeneity would be assumed within each of the containers, which are estimated to number 53,400 (Raivo et al. 1993). This approach would be the most exact, but the work required would be far beyond the scope of this study. In addition, detailed container data are not available for much of the waste.

Several options exist between these two extremes. The intermediate options involve varying degrees of simplification and categorization. In one option, the untreated ALLW could be modeled by assuming homogeneity of all waste containers within each of the approximately 90 *content codes* that are in use for this waste (Raivo et al. 1993). (Each container of waste stored at the INEL is categorized under a specific content code, which provides a general description of the containerized waste.) Actually, the waste is not always similar from one container to another within a content code. For example, content code 111 includes Waste Isolation Pilot Plant (WIPP) certified solid wet sludge from Rocky Flats Plant and laboratory research waste from Argonne National Laboratory-East (ANL-E) (Apel et al. 1993). Thus, even the use of about 90 content codes involves grouping containers whose contents vary to some extent. And 90 groups

of waste is still far too many to use in a parametric performance assessment (PA) study requiring multiple computer runs.

PSPI documents (e.g., Raivo et al. 1993) group the ALLW by *INEL identification (ID) number*. For mixed ALLW, 18 INEL ID numbers exist—groups of content codes—that are based primarily on physical similarities. Examples are cemented sludges (six content codes), resins (three), metals (five), combustibles (seven). For nonmixed ALLW, there are 8 similar ID numbers.

The generic descriptors for the waste groupings by ID number should not be taken literally. For example, the name of ID number ID-EGG-144 is "radioactive sources." It includes three content codes, all of which are remote-handled waste. However, only one of the three content codes involves waste commonly thought of as radioactive sources, i.e., a concentrated source of radioactivity that is inside a metal container and is used for radiation applications. The other two content codes involve waste commonly considered combustibles or laboratory equipment.

The waste groupings by INEL ID number were used as the starting point for the present refined modeling of the untreated waste. The INEL ID numbers were then grouped into a smaller number of waste *categories*. The characteristics of the waste within each category were then evaluated and compared against the characteristics of the other categories.

Waste records in the form of detailed descriptions of the waste within a content code (Apel et al. 1993, supplemented by Apel et al. 1994 as necessary) were used to arrive at appropriate categories for modeling the waste. Those descriptions provide a large amount of information on the waste, although they are incomplete for many content codes, particularly concerning the radionuclide activities and distributions.

B-3. SOURCE OF INFORMATION FOR ACTINIDE CONTENT

To develop the actinide content of the untreated ALLW, the initial plan was to use the histograms in Apel et al. 1993 that showed the number of containers vs. transuranic (TRU) activity. Those histograms cover the range of TRU activity from 0 to 100 nCi/g in decades (e.g., 0–10, 10–20, 20–30). Apel et al. 1994 eliminated the histograms for the ALLW.

The histograms in Apel et al. 1993 were examined in this study to determine if certain types of ALLW tended to be higher than others in average TRU activity. However, few instances of such a tendency were observed. Most of the distributions showed little tendency of waste packages to concentrate in the top or bottom part of the 0 to 100 nCi/g domain. Rather, most distributions showed a scatter across the domain. In addition, when the content codes were grouped under ID numbers, the resulting distributions were found to be rather similar from one ID number to another. Finally, radioactivity information was missing for a large number of the content codes, so no histograms were available for these content codes. Therefore, the histograms were not used in determining whether one waste category has a higher concentration of TRU activity than another category.

Rather than using the histograms in Apel et al. 1993, the total masses of actinides for each content code (*where known*) were used as an *aid* in comparing the actinide activity among content codes, ID numbers, and categories. The total masses of actinides are listed in Part C of Table B-5 and Part B of Table B-6 of Raivo et al. 1993. Those data were used only for scaling the actinide content among content codes, ID numbers, and categories. *The actinide inventory used was that developed in Appendix A of the present report, as adapted for the untreated waste.* See Section B-6 below.

B-4. SOURCE OF INFORMATION FOR NONACTINIDE CONTENT

Neither Raivo et al. 1993, nor Apel et al. 1993, provided information on nonactinide content. This was true even for the remote-handled (RH) waste, the high radiation levels of which must be a result of significant activity of nonactinides. *The approach used to estimate the nonactinide activity was that developed in Section A-9 of Appendix A, as adapted for the untreated waste.* See Section B-6 below.

B-5. GROUPING THE WASTE BY CATEGORY

B-5.1 Overview

Table B-1 at the end of this appendix indicates the process of grouping the waste into three categories. (The information in Table B-1 was taken primarily from Raivo et al. 1993 and Apel et al. 1993. Apel et al. 1994, which was completed when this study was already underway, was consulted in instances in which the information could not be found in the other two sources.) In the PA computer runs performed for each category, homogeneity was assumed within a category.

Column 1 of Table B-1 lists the three categories into which the ALLW was grouped: sludges, etc.; nonsludges; and remote-handled waste. The rationale for these categories will be explained in Section B-5.3.

Column 2 lists the INEL ID numbers for the waste, e.g., ID-EGG-112. Columns 3 and 4 list the names (brief descriptions) and the content codes comprising each ID number. ID numbers for mixed ALLW and nonmixed ALLW (those whose ID numbers end with an "N") were combined into the same entry. The content codes for nonmixed ALLW are listed after those for the mixed ALLW, separated by a semicolon.

Column 5 indicates whether the waste in the ID number is contact-handled (CH) or remote-handled (RH). Column 6 lists the volume of the waste for each ID number, in m³.

Column 7 indicates the quantity of actinides in the ID number. Lumping the actinides together in this manner was a simplification necessitated by the nature of this study and was done for scaling purposes only. In reality, different content codes have different mixtures of actinides. For some content codes, all the actinides are listed as Pu-52, the standard weapons-grade Pu mixture consisting of about 94% Pu-239 by mass, with small amounts of Pu-238, 240, 241, and 242. Other content codes are listed as having Pu-52 plus some Pu-239. Such entries are incomplete because Pu-239 is not found without some accompanying Pu-240. Some content

codes are listed with Pu-52 plus U-235. Again, in reality, the U-235 always has some accompanying U-234 and U-238, but the records are incomplete on this point. For many content codes, no information on the actinide content is available in the records. Because the records are incomplete in several ways, no fully defensible breakdown of actinides is possible until the waste is removed from storage and assayed. Thus, lumping the actinides together, *for purposes only of scaling the actinide activity in the various categories*, is considered of sufficient accuracy for this study.

The 8th column indicates what the records list for the nonactinides in the waste. No such information was found, so the nonactinides were estimated using the approach developed in Section A-9 of Appendix A, as adapted to the untreated ALLW.

The last column indicates the mechanism, on an overall basis, for modeling the release of radionuclides for each ID number.

B-5.2 Waste Termed "Uncategorized Waste"

The waste in ID number 287, which is called "uncategorized waste," represents a special case. This waste consists of (a) 137 m³ of content code 0 waste, which was buried at the Subsurface Disposal Area (SDA) (mostly in 1970), later retrieved, and placed into storage; and (b) 2,006 m³ of content code 9999 waste,^a which was received from waste generators and placed into storage from 1970 to 1972, before definitive waste information was developed by the generators. The total waste volume for this ID number is almost 10% of the total ALLW. Little information is available on the waste, particularly the 1970–1972 stored waste. Therefore, the average characteristics for the waste in ID number 287 (which was generated mostly from 1970 through 1972) were assumed here to match those of waste generated after 1972.

The waste in ID number 287 was then divided into two portions. One portion was assumed to match the type of waste in category 1, the sludges, etc. The other portion was assumed to match the type of waste in category 2, the nonsludges. None of the waste in ID number 287 was listed as RH, so none of it was assumed to match the type of waste in category 3.

For the waste in ID number 287, the volumetric split and the actinide inventory split between categories 1 and 2 were made so as to match the splits for the later waste, as exhibited in Table B-1. Excluding ID number 287, the volume of category 1 waste was 3,561 m³; for category 2 waste, 18,656 m³. Therefore, the 2,142 m³ volume of ID number 287 waste was divided into $2,142 \text{ m}^3 \times 3,561 / (3,561 + 18,656) = 343 \text{ m}^3$ sludges and $2,142 \text{ m}^3 \times 18,656 / (3,561 + 18,656) = 1,799 \text{ m}^3$ of nonsludges.

The actinide inventory listed in the records for ID number 287 is 15 g. (This value may be quite low because there are no data on actinide content for the 1970–1972 waste placed into storage. The records implicitly assume 0 g for the actinides in this waste.) The 15 g was divided

a. Apel et al. 1994 revised the estimated volume to 3,008 m³ after much of the PA work had been completed. No significant effect of the revision on the PA results would be expected. For consistency with the earlier PA work reported in this document, the revised volume was not used.

based on the relative fractions of the total actinides in the two categories, not counting ID number 287. Those values are about 568 g for category 1 and 2,892 g for category 2. Accordingly, 2 g of the actinides in ID number 287 were apportioned to category 1 and 13 g to category 2. The totals become about 570 g for category 1 and 2,905 g for category 2.

B-5.3 Rationale for the Categories

The first column of Table B-1, the categories, is now discussed in more detail.

The first 5 ID numbers relate to waste in the form of sludges, resins, and salts. They are combined into category 1: sludges, etc. The first 3 ID numbers are expected to contain more than 99% of the actinides in the first 5 ID numbers, and their expected release mechanism is diffusion. The next 2 ID numbers have expected release mechanisms of desorption and dissolution, respectively, but their actinide inventories are negligible by comparison with that of the first three ID numbers. Thus, the first 5 ID numbers were modeled as sludges with release parameters matching those for content code 2, second-stage sludge, the dominant content code in terms of actinide inventory. The 6th entry is the sludge portion of ID number 287. For scaling purposes only, the total actinide content of category 1 was taken to be 570 g, and the total volume was taken to be 3,904 m³. The release mechanism was assumed to be diffusion.

All but one of the remaining ID numbers were grouped into category 2, the nonsludges. Most of the actinides in category 2 are located on the surfaces of paper, rags, and metals. For all of these ID numbers except 141 and 288, the expected release mechanism is surface washoff. ID number 141 includes about 894 g of U-235 in U₃O₈ pellets, with a dissolution release mechanism. However, the activity involved is much less than 1 Ci. ID number 288, which is mostly U-235 and again if of very low activity, includes about 27 g of actinides. For scaling purposes only, the actinide content of category 2 was assumed to be 2,905 g, and the volume was assumed to be 20,455 m³. The release mechanism was taken to be surface washoff.

The last ID number is 144, which contains all of the RH ALLW. The waste with this ID number was assigned to category 3, the RH waste. The volume is 31 m³, chosen so as to match the volume used elsewhere in this report and in the Radioactive Waste Management Information System (RWMIS). Information on the actinide content is inconclusive. A total of 560 g is identified in the reports, but that value is incomplete. The information on physical and chemical form is not detailed; a surface washoff release mechanism was assumed because most of the waste is combustibles and (apparently) nonactivated metals.

B-5.4 Conclusions about Radiological Characteristics

The Table B-1 entries for the category rollups can be used to draw some conclusions about the radiological characteristics of the waste.

One of the key questions is whether the concentrations of radionuclides differ between the two CH categories, i.e., categories 1 and 2. Within the accuracy of this study, the concentrations of actinides (by volume) are essentially the same for both categories. The actinide concentration is $570 \text{ g}/3,904 \text{ m}^3 = 0.15 \text{ g/m}^3$ for category 1 and $2,905 \text{ g}/20,455 \text{ m}^3 = 0.14 \text{ g/m}^3$ for category 2.

Table B-1. Categorization of the untreated ALLW.

Category	INEL ID no.: ID-EGG ^a	Name	Content codes ^b	CH or RH?	Vol (m ³)	Quantity of actinides (from Raivo et al. 1993, Apel et al. 1993, and, as necessary, Apel et al. 1994) for purposes of scaling only	Nonactinide activity (from information sources at left)	Assumed release mechanism
1- Sludges, etc.	112, 112N	Cemented sludges	4, 112, 114, 834, 835, 976; 836, 978	CH	1,301	c. 63 g; most is in content code 4 (special setups)	None listed	Diffusion
	102, 102N	Uncemented inorganic sludges	1, 2, 7, 111, 290, 292; 30, 95, 113, 995	CH	1,233	c. 365 g; most is in content code 2 (second stage sludge) and content code 7 (Bldg. 347 dry sludge)	None listed	Diffusion
	158	Uncemented organic sludge	3	CH	1,002	c. 140 g	None listed	Diffusion
	145	Resins	430, 431, 432	CH	11	<1 g	None listed	Desorption
	146	Salts	5	CH	14	<1 g	None listed	Dissolution
	287 (part—see text for explanation)	Uncategorized	0 ^c , 9999	CH	343	c. 2 g in content code 0 (waste retrieved from burial). No data on content code 9999 (waste received 1970–1972)	None listed	Diffusion
		Total/Composite: Sludges, etc.		CH	3,904	c. 570 g	None listed; developed using approach in Appendix A	Diffusion using parameters for content code 2 waste

Table B-1. (continued).

Category	INEL ID no.: ID-EGG- ^a	Name	Content codes ^b	CH or RH?	Vol (m ³)	Quantity of actinides (from Raivo et al. 1993, Apel et al. 1993, and, as necessary, Apel et al. 1994) for purposes of scaling only	Nonactinide activity (from information sources at left)	Assumed release mechanism
2- Non- sludges	109	Benelex, plexiglas	302, 464	CH	59	c. 8g.	None listed	Surface washoff
	114, 114N	Combustibles	116, 330, 336, 337, 826, 900, 970; 847	CH	5,674	c. 1,223 g; most is in content code 330 (dry paper and rags)	None listed	Surface washoff
	115, 115N	Concrete-brick	122, 371, 375; 960	CH	880	c. 4g	None listed	Surface washoff
	118	Filters	119, 158, 328, 335, 338, 360, 376, 490	CH	1,347	c. 55g; most is in content code 490 (high-efficiency particulate air and other filters)	None listed	Surface washoff
	119, 119N	Glass	440, 441, 442; 118	CH	405	c. 1g	None listed	Surface washoff
	120	Glovebox gloves	123, 339, 463	CH	8	< 1g	None listed	Surface washoff
	132	Metals	117, 320, 480, 481, 803	CH	7,031	c. 410g; most is in content code 480 (gloveboxes, furnaces, lathes, ducting, etc.)	None listed	Surface washoff
	134, 134N	Miscellaneous	100, 101, 121, 201, 241, 950; 203, 824, 825	CH	2,453	c. 270g; most is in content code 100 (general plant waste) and content code 201 (noncombustible solids)	None listed	Surface washoff

Table B-1. (continued).

Category	INEL ID no.: ID-EGG- ^a	Name	Content codes ^b	CH or RH?	Vol (m ³)	Quantity of actinides (from Raivo et al. 1993, Apel et al. 1993, and, as necessary, Apel et al. 1994) for purposes of scaling only	Nonactinide activity (from information sources at left)	Assumed release mechanism
2- Non-sludges (cont.)	137, 137N	Nonmetal molds and crucibles	300, 301; 115, 310, 370	CH	26	< 1g	None listed	Surface washoff
	141, 141N	Particulate wastes	90, 150, 372, 374, 990; 425, 842	CH	588	c. 894g; most is U-235 in content code 150 (miscellaneous waste, including normal and enriched U ₃ O ₈ pellets) ^d	None listed	Dissolution
	287 (part—see text for explanation)	Uncategorized	0 ^c , 9999	CH	1,799	c. 13 g in content code 0 (waste retrieved from burial). No data on content code 9999 (waste received 1970–1972)	None listed	Surface washoff
	288	Unknown	10, 20, 102, 155, 156, 161, 202, 204, 838, 848	CH	185	c. 27 g; most is U-235 in content code 102 (absorbed liquids)	None listed	Desorption
		Total/Composite: Nonsludges		CH	20,455	c. 2,905 g; detailed distribution developed using approach in Appendix A.	None listed; developed using approach in Appendix A	Surface washoff

Table B-1. (continued).

Category	INEL ID no.: ID-EGG- ^a	Name	Content codes ^b	CH or RH?	Vol (m ³)	Quantity of actinides (from Raivo et al. 1993, Apel et al. 1993, and, as necessary, Apel et al. 1994) for purposes of scaling only	Nonactinide activity (from information sources at left)	Assumed release mechanism
3- Remote- handled	144	Radioactive sources	104, 153, 154	RH	31 ^c	No data in Raivo et al. 1993 or Apel et al. 1993. Apel et al. 1994 lists c. 560 g (< < 1Ci), almost entirely U-235 and U-238, in content codes 104 (miscellaneous combustible and noncombustible alpha hot cell waste—the dominant content code by volume) and 153 (combustibles with fuel on filters—the dominant content code by actinide content). No data on content code 154, but it contains Pu-Be sources, a Ra needle, and experimental fuel capsules	None listed	Highly varied, little information available.
		Total/composite: RH Waste		RH	31 ^c	More than 560 g; detailed distribution developed using approach in Appendix A.	None listed; developed using approach in Appendix A	Surface washoff
		Grand total			24,390 ^c			

NOTES:

a. INEL ID Numbers ending in N represent ALLW that is nonmixed. All other entries represent mixed ALLW.

b. The content codes for any nonmixed ALLW are listed last in each cell of this column, after the semicolon.

c. For consistency with other analyses in this document, the RH waste volume was assumed to be 31 m³, as given in Appendix A, rather than the 54 m³ given in Raivo et al. 1993. The total volume over all categories is, therefore, 23 m³ less than the 24,413 m³ in Raivo et al. 1993. This difference of less than 0.1% is not significant for developing the radiological inventory and was neglected.

d. Interestingly, the ALLW records do not list the presence of U-238 for this content code, even though it must be present if pellets of normal (unenriched) U₃O₈ are present. This is one example of the incompleteness of the radiological information about the waste.

This result substantiates the simplifying assumption made in Appendix A that the actinide activity in the CH ALLW is evenly spread. Thus, little accuracy would be gained by using two different concentrations of actinides for the CH ALLW in the two categories.

The actinide distribution developed in Appendix A (see Table A-3) was used for all categories of untreated ALLW, apportioned according to the relative volumes of the three categories. It is consistent with the actinide distribution used for the treated waste.

For the nonactinide distribution, the ALLW records provide no information. Therefore, the refined actinide distribution developed in Appendix A (see Tables A-8 and A-9) was used for the untreated ALLW.

B-6. FINAL RESULTS FOR REFINED INVENTORY MODEL

Table B-2 provides the refined inventory model for the untreated waste. It is based on the categories, volumes, release mechanisms, and key content codes of Table B-1, supplemented by information developed in Appendix A on the radionuclide distributions.

The radionuclide distributions were developed as follows. For category 3 (RH waste), the nonactinide values were taken from Table A-9 in Appendix A on RH ALLW. The actinide values were taken from Table A-3 of Appendix A, then multiplied by the volume fraction of category 3 waste to all the ALLW (31/24,413). (The minute fractional difference of less than 0.1% between the total volume of 24,413 m³ in Appendix A and the total volume of 24,390 m³ in Table B-1 was neglected.)

For categories 1 and 2 (CH waste), the approach was similar. Nonactinide values were taken from Table A-8 in Appendix A on CH ALLW, then multiplied by the fraction of CH volume that is category 1 (3,904/24,359) or category 2 (20,455/24,359). The actinide values were taken again from Table A-3 of Appendix A, then multiplied by the volume fraction of category 1 or 2 waste to all the ALLW (3,904/24,413 or 20,455/24,413, respectively).

Table B-2. Refined model of the untreated ALLW.

Category	Volume (m ³)	Actinide inventory (Ci)	Nonactinide inventory (Ci)	Release mechanism	Content code for other properties
1- Sludges, etc.	3,904	Am-241 214 Cm-244 1.28 Pu-238 145 Pu-239 95.7 Pu-240 23.8 Pu-241 384 U-233 2.08 U-238 0.0003	Ba-137m 8.70 Co-60 4.74 Cs-137 8.70 Sr-90 4.95 Y-90 4.95	Diffusion	2 (second-stage sludge)
2- Nonsludges	20,455	Am-241 1,124 Cm-244 6.71 Pu-238 760 Pu-239 502 Pu-240 123 Pu-241 2,014 U-233 10.9 U-238 0.0014	Ba-137m 45.6 Co-60 24.9 Cs-137 45.6 Sr-90 25.9 Y-90 25.9	Surface washoff	330 (dry paper and rags)
3- Remote-handled	31 ^a	Am-241 1.647 Cm-244 0.010 Pu-238 1.15 Pu-239 0.735 Pu-240 0.181 Pu-241 2.950 U-233 0.016 U-238 0	Ba-137m 882 Co-60 6.8 Cs-137 882 Sr-90 798 Y-90 798	Surface washoff	None appropriate; assume particulate on filters or rags
Total	24,390 ^a	Am-241 1,340 Cm-244 8 Pu-238 906 Pu-239 598 Pu-240 147 Pu-241 2,401 U-233 13 U-238 0.0017	Ba-137m 936 Co-60 36.4 Cs-137 936 Sr-90 829 Y-90 829	NA	NA

a. For consistency with other parametric studies in this document, the volume of the RH waste was assumed to be 31 m³, as given in RWMIS and discussed in Appendix A, rather than the 54 m³ given in Raivo et al. 1993. The total volume over all categories is, therefore, 23 m³ less than the 24,413 m³ in Raivo et al. 1993. This volume difference of less than 0.1% is not significant for developing the radiological inventory and was neglected.

REFERENCES FOR APPENDIX B

- Apel, M. L., M. T. Muneta-Bradford, A. L. Rubert, J. C. Tozer, M. D. Argyle, and G. K. Becker, 1993, *Waste Description Information Package for Alpha Low-Level Mixed Waste & Alpha Low-Level Waste (draft)*, Attachment 2 to *Solid Alpha-Contaminated Low-Level Radioactive Waste Transport/Treatment Services Request for Proposal*, SP-SOW-01, July.
- Apel, M. L., G. K. Becker, Z. K. Ragan, J. Frasure, B. D. Raivo, L. G. Gale, and D. P. Pace, 1994, *Radiological, Physical, and Chemical Characterization of Low-level Alpha-contaminated Wastes Stored at the Idaho National Engineering Laboratory for Use in Conducting Feasibility Studies for Treatment Services*, EG&G Idaho, EGG-RWMC-11189, March.
- Raivo, B. D., T. L. Eddy, and G. L. Anderson, 1993, *Statement of Work: Commercial Transport and Treatment of Alpha-Contaminated Mixed Low-Level Radioactive Waste (draft)*, Attachment 1 to *Solid Alpha-Contaminated Low-Level Radioactive Waste Transport/Treatment Services Request for Proposal*, SP-SOW-01, June.

Appendix C

Completed Questionnaires Used to Obtain Geologic and Hydrologic Properties of Disposal Locations A, B, and C

Appendix C

Completed Questionnaires Used to Obtain Geologic and Hydrologic Properties of Disposal Locations A, B, and C

This appendix contains the completed questionnaires used to select the geologic and hydrologic properties representative of disposal locations A, B, and C. References cited by the persons who completed the questionnaires are not included in this appendix.

The properties selected for disposal locations D and E were obtained from formal performance assessments (PAs) prepared recently by personnel at those respective sites. Those formal PAs are referenced in Section 3 of this report.

**Site Properties Representative of Site A
(Radioactive Waste Management Complex at the INEL)**

Table D-3. Parameters used in the ground water impact calculations.

Parameter	Value	Units	Reference
Source length parallel to flow	257	m	Case et al. (1990)
Source width parallel to flow	122	m	Case et al. (1990)
Source thickness	6.1	m	Case et al. (1990)
Soil density at source	1.5	g/cm ³	Case et al. (1990)
Aquifer density	1.9	g/cm ³	Case et al. (1990)
Aquifer porosity	0.10	-	Case et al. (1990)
Pore velocity	570	m/yr	Case et al. (1990)
Well screen thickness	12	m	ES&G Idaho (1991)
Unsaturated zone thickness	176	m	Case et al. (1990)
Infiltration rate	0.123	m/yr	Assumed ^a
Source volumetric water content	0.347	-	Calculated ^b
Unsaturated volumetric water content	0.07	-	Calculated ^b
Longitudinal dispersivity	0.9	m	Arnett et al. (1990) ^c
Transverse dispersivity	0.4	m	Arnett et al. (1990) ^c
Receptor distance	100.0	m	- ^d
Water ingestion rate	2.0	L/day	CFR (1991)

a. Based on a 100-year water travel time in the unsaturated zone. See discussion in Section 4.2.2.3.

b. Based on the 100-year water travel time in the unsaturated zone.

c. Value divided by 100 for conservatism.

d. The value of 100 m is generally accepted as the point of compliance for performance assessment calculations for DOE facilities. This distance is measured from the downgradient edge of the facility.

Table D-6. Parameters used in the ground water impact calculations (values different from Table D-3 are shown in bold).

Parameter	Value	Units	Reference
Source length parallel to flow	257	m	Case et al. (1990)
Source width parallel to flow	122	m	Case et al. (1990)
Source thickness	6.1	m	Case et al. (1990)
Soil density at source	1.3	g/cm ³	Case et al. (1990)
Aquifer density	1.9	g/cm ³	Case et al. (1990)
Aquifer porosity	0.10	-	Case et al. (1990)
Pore velocity	570	m/yr	Case et al. (1990)
Well screen thickness	12	m	EG&G Idaho (1991)
Unsaturated zone thickness	176	m	Case et al. (1990)
Infiltration rate	0.123	m/yr	Assumed ^a
Source volumetric water content	0.347	-	Calculated ^b
Unsaturated volumetric water content	0.07	-	Calculated ^b
Longitudinal dispersivity	9.0	m	Arnett et al. (1990) ^c
Transverse dispersivity	4.0	m	Arnett et al. (1990) ^c
Receptor distance	100.0	m	- ^d
Water ingestion rate	2.0	L/d	40 CFR 141

a. Based on a 100-year water travel time in the unsaturated zone. See discussion in Section 4.2.2.3.

b. Based on the 100-year water travel time in the unsaturated zone.

c. These values were reduced by a factor of 10 for scale considerations.

d. The value of 100 m is generally accepted as the point of compliance for performance assessment calculations for DOE facilities. This distance is measured from the downgradient edge of the facility.

**Site Properties Characteristic of Site B
(Site 14 in the Central Part of the INEL)**

PostNet® Fax Transmittal Form 7571 • of 2 pages • 4	
To: Jonathan Myers	From: Jack Barracough
Co: I. T. Corp.	Co: EG&G Idaho
City: Albuquerque	Phone: 208-526-1887
Fax: 505-262-8855	Fax: 208-526-0875

May 25, 1993

RECEIVED

Project No. 3C1624 02

JUN 01 1993

IT CORP. - ALBUQUERQUE

Mr. Jack Barracough
EG&G Idaho, Inc.
P. O. Box 1625
Idaho Falls, ID 83415

Request for INEL Site-14 Hydrological Data

Dear Mr. Barracough:

The Idaho National Engineering Laboratory (INEL) is planning long-term disposal of mixed low-level radioactive waste. The waste will be treated to satisfy the requirements of the Resource Conservation and Recovery Act, and a performance assessment of the disposal system will be performed to ensure compliance with applicable regulations governing the fate and transport of radioactive contaminants.

As part of ongoing support to EG&G Idaho, IT Corporation will be evaluating three potential final waste forms at two arid disposal sites (INEL and NTS) in terms of compliance with the performance objectives in DOE Order 5820.2A and other applicable regulations. As per Tom Smith's request, we have prepared a questionnaire requesting a number of site properties for Site-14 at the INEL. The parameters that are marked with a check mark are the critical data needed. Please send the completed questionnaire by FAX to (505) 262-8855 by Thursday, May 27, if possible.

Please feel free to call me at (505) 262-8726 or Sinisa Djordjevic at (505) 262-8746 if you have any questions. Thanks in advance for your assistance in providing this information.

Sincerely,

IT CORPORATION

Jonathan Myers

JM:SD:lrg

cc: Sinisa Djordjevic, IT Corporation
Tom Smith, EG&G Idaho, Inc.

Regional Office

3301 Central Avenue, N.E. • Suite 700 • Albuquerque, New Mexico 87108 • (505) 262-8800

IT Corporation is a wholly owned subsidiary of International Technology Corporation

Questionnaire on Site Hydrological Data

Contact Information

Person Completing this Questionnaire:

Name: Jack T. Barracough, EG & G Idaho, Inc.

Title: Scientific Specialist - Hydrologist

Site (e.g., Hanford, INEL, ORNL): INEL

Address: EG & G Idaho, Inc. P.O. Box 1625

City: Idaho Falls State: Idaho ZIP Code: 83415-2107

Telephone: (208) 526-1887 FAX: (208) 526-0875

Date (month/day): 5 128 193

Parameter	Median (Expected Value)	Distribution Type	Distribution Parameters		Units	Reference
Precipitation rate	20.42		MAX	MIN	cm/yr	Climateograph of DEN by NOAA
<u>MAX to October</u> Class A pan evaporation rate	104.1		38.1	12.22	cm/yr	NOAA Tech. Rep. no. 93
Infiltration rate (if measured)	—		—	—	—	USGS open-file Report 84-231
Thickness of the unsaturated zone (depth to groundwater)	80.47		0.45	0.15	m	WINCO-1042
Unsaturated zone porosity	0.3		0.15	0.05	—	
<u>Section 1</u> <u>Basalt</u>	0.7					

INEL - SITE 14 - T5N, R31E, 28cc1

Parameter	Median (Expected Value)	Distribution Type	Distribution Parameters		Units	Reference
			MAX	MIN		
Unsaturated zone residual moisture content	0.06		0.85	0.03	—	Estimated based on IDO-22056
Unsaturated zone saturated hydraulic conductivity	3.0E-04		9.1E-04	1.10E-04	cm/sec	Estimated based on IDO-22056
Unsaturated zone relative permeability curve (please attach if available)	3.5E-03		3.5E-02	3.5E-04	cm/sec	
Aquifer groundwater velocity	668		1223	332	m/yr	Estimated from IDO-22053
Aquifer gradient	1.44E-03		1.58E-03	1.35E-03	—	IDO-22048
Aquifer hydraulic conductivity	2.58E-01		4.70E-01	3.92E-02	cm/sec	IDO-22034
Aquifer effective porosity	0.1		0.15	0.03	—	IDO-22054
Aquifer thickness	366		457	91	m	nearby deep wells
Aquifer longitudinal dispersivity	90		130	50	m	IDO-22054
Aquifer transverse dispersivity	40		140	20	m	IDO-22054
Distance to surface water body in hydraulic linkage with the aquifer	215				km	Down gradient to the Snake River
Distance to receptor well	12.1				km	to well Site C.
Average water flow rate in the surface water body	—					No surface water

INEL - SITE 14 - TSN, R31E, 28CCJ

Parameter	Median (Expected Value)	Distribution Type	Distribution Parameters		Units	Reference
			α MAX	λ MIN		
Surface-water downstream distance from release point to withdrawal location	—					No release to surface water
Average water depth of the surface-water body	2.13		3.66	0.91	m	Snake River
Average width of the surface- water body	122		145	84	m	Snake River

Distribution Types:

Constant. When a distribution type is listed as constant, a distribution has not been assigned and a constant value is used in all calculations.

Unknown. If the distribution is unknown but a range is known then α is the lower value of the range and λ is the upper value of the range.

The distribution can be described by any one of two continuous probability density functions (pdf) as follows:

Beta. Beta designates the beta pdf. This functional form of the pdf is given by:

$$f(x) = \frac{1}{B(\alpha, \lambda)} \frac{(x-a)^{\alpha-1} (b-x)^{\lambda-1}}{(b-a)^{\alpha+\lambda-1}}$$

**Site Properties Characteristic of Site C
(Radioactive Waste Management Site at the Nevada Test Site)**

**REYNOLDS ELECTRICAL &
ENGINEERING CO., INC.**

Post Office Box 98521
Las Vegas, NV 89193-8521

FACSIMILE MESSAGE

UNCLASSIFIED ONLY

THIS COVER SHEET IS FOR THE TRANSMITTAL
OF UNCLASSIFIED MESSAGES ONLY

DATE: 5/27

TIME: 8:15

TO: (505) 262-8855
FACSIMILE PHONE NO.

TO: _____
VERIFICATION PHONE NO.

TO: SINISA DJORDJEVIC / JONATHAN MYERS
NAME ORGANIZATION/GEOGRAPHICAL LOCATION OFFICE PHONE NO.
IT CORP, ALB.

FROM: MAX DOLENC, REECO (702) 295-584
NAME DEPARTMENT TELEPHONE NO.

THIS TRANSMITTAL
CONSISTS OF 6 PAGES (EXCLUDING COVER SHEET)

SPECIAL INSTRUCTIONS:

UNCLASSIFIED ONLY

2 E6+G/SAIC, 1991 Radioactive waste performance assessment for the Area
Radioactive Waste Management Site at the Nevada Test Site. E6+G Idaho, Inc. and
Sevens Applications, Environmental Corporation (SAIC), Idaho Falls.

Questionnaire on Site Hydrological Data

Contact Information

Person Completing this Questionnaire:

Name: Doc Hammarosta / Dan Blout

Title: Section Chief / Hydrogeologist

Site (e.g., Hanford, INEL, ORNL): Nevada Test Site, Area 5 RWMS

Address: REEL, MS. 738, P.O. Box 754, Las Vegas NV

City: Las Vegas

State: NV

ZIP Code: 89193-8521

Telephone: (702) 295-6408 / 7435

FAX: (702) 295-7435

Date (mm/dd/yy): 5/25/93

Parameter	Mean Median (Expected Value)	Distribution Type	Distribution Parameters		Units	Reference
			α	λ		
Precipitation rate	10	NA			cm/yr	ES&G, SAIC PA 2
Class A pan evaporation rate	220	NA			cm/yr	ES&G, SAIC PA 2
Infiltration rate (if measured)	781	TD				
Thickness of the unsaturated zone (depth to groundwater)	805	NA	770	840	ft	Pilot Well Report
Unsaturated zone porosity	37.8	TD	76.0	55.9	percent	Pilot Well Report

Parameter	Median (Expected Value)	Distribution Type	Distribution Parameters		Units	Reference
			α	λ		
Unsaturated zone residual moisture content	5.5	TBD	0.0	10.8	percent	Pilot Well Report
Unsaturated zone saturated hydraulic conductivity	2.8×10^{-3}	TBD	2.1×10^{-5}	5.8×10^{-3}	cm/sec	Pilot Well Report
Unsaturated zone relative permeability curve (please attach if available)		N/A				Pilot Well Report
Aquifer groundwater velocity	N/A	N/A				Pilot Well Report
Aquifer gradient	N/A	N/A				Pilot Well Report
Aquifer hydraulic conductivity	1.0×10^{-3}	TBD	2.4×10^{-4}	1.9×10^{-3}	cm/sec	Pilot Well Report
Aquifer effective porosity	TBD	TBD				
Aquifer thickness	TBD	TBD				
Aquifer longitudinal dispersivity	TBD	TBD				
Aquifer transverse dispersivity	TBD	TBD				
Distance to surface-water body in hydraulic linkage with the aquifer	N/A	N/A				
Distance to receptor wet						
Average water flow rate in the surface-water body	N/A	N/A				

Parameter	Median (Expected Value)	Distribution Type	Distribution Parameters		Units	Reference
			α	λ		
Surface-water downstream distance from release point to withdrawal location	N/A	N/A				
Average water depth of the surface-water body	N/A	N/A				
Average width of the surface- water body	N/A	N/A				

Distribution Types:

Constant. When a distribution type is listed as constant, a distribution has not been assigned and a constant value is used in all calculations.

Unknown. If the distribution is unknown but a range is known then α is the lower value of the range and λ is the upper value of the range.

The distribution can be described by any one of five continuous probability density functions (pdf) as follows:

Beta. Beta designates the beta pdf. This functional form of the pdf is given by:

$$f(x) = \frac{1}{B(\alpha, \lambda)} \frac{(x-a)^{\alpha-1} (b-x)^{\lambda-1}}{(b-a)^{\alpha+\lambda-1}}$$

Figure 3.13 Fitted unsaturated hydraulic conductivity functions for core samples from Ue5PW-1.

Figure 3.16 Fitted unsaturated hydraulic conductivity functions for core samples from UeSPW-2.

Figure 3.17 Fitted unsaturated hydraulic conductivity functions for core samples from UaSPW-3.

Appendix D

Description of Preprocessor and Postprocessor Programs

Appendix D

Description of Preprocessor and Postprocessor Programs

D-1. MLLWDRIV.FOR

D-1.1 Description

MLLWDRIV.FOR is a driver preprocessor program, written in FORTRAN, that reads the input file for the Model and creates the appropriate input data files for all the codes in the Model. This program also links the individual codes or modules into the Model as explained in this section. The program first opens the input data file "MLLWDRIV.DAT" and reads in the values of the variables for a given run. The data encompass waste form, disposal facility, site, and environmental properties.

After reading the variables in the file "MLLWDRIV.DAT," the file "RAD.DAT" is created for use by the "WELLC.FOR" code. The parameters that are written to the "RAD.DAT" file include the number of radionuclides to be modeled, the element symbol identifiers, the atomic weights of the nuclides, and the nuclide retardation coefficients.

The program then creates the data file "RADUST.DAT" containing data needed to create the DUST input file using the PREDUST.FOR preprocessor. Specifically, the parameter values that are written to the RADUST.DAT file include the number of radionuclides to be simulated, the radionuclide identifiers required by the DUST code, the radionuclide half-lives (yr), the atomic weights, the saturation concentrations (g/cm^3), and the radionuclide concentrations in the waste form (Ci/m^3).

The program MLLWDRIV then calls the subroutine VDTTHETA, passing to this subroutine the value of the infiltration rate into the disposal facility, denoted by variable RATEINFL. The subroutine VDTTHETA passes back the Darcy velocity and the moisture content in the facility. Program MLLWDRIV then creates the data file required by the program PREDUST.FOR, which in turn creates the input data file for each radionuclide for the DUST code. This will be explained later in this Appendix.

The data file for the WELLC code (PREWELLC.IN) is then created. The final statements in program MLLWDRIV create the DOS batch file "PAM.BAT," which, when executed, runs the Model. Since the DUST code can calculate the source term (time-dependent flux of a radionuclide leaving the disposal facility) for only one radionuclide at a time, the DUST code must be executed once for each radionuclide. Thus, a DO loop for the number of radionuclides writes statements to the batch file to cause execution of the PREDUST, DUST, and POSTDUST codes for the required number of radionuclides. Statements are then written to the PAM.BAT batch file to cause execution of the WELLC code. A statement is then written to the batch file that causes execution of the postprocessor DECAYDOS that decays and in-grows radionuclides at the water well used by the receptor.

D-1.2 Verification

The program MLLWDRIV.FOR is a peripheral computer code that manipulates the data read in from the master input data file (MLLWDRIV.DAT) and creates the appropriate input data files for the various codes and pre- and postprocessors of the Model.

Verification of the code was demonstrated by comparing the data in MLLWDRIV with the following input data files:

- PREDUST.DAT
- RAD.DAT
- RADUST.DAT
- PREWELLC.IN
- PAM.BAT.

The comparison of these files was successful and ensures that execution of the MLLWDRIV code produces the required input data files, thus verifying the MLLWDRIV code.

D-2. PREDUST.FOR

D-2.1 Description

PREDUST.FOR is a preprocessor program of the Model written in FORTRAN that prepares the input data file for the DUST code for each radionuclide analyzed. Since the DUST code can calculate the source term (time-dependent flux of a radionuclide leaving the disposal facility) for only one radionuclide at a time, the DUST code must be executed once for each radionuclide. This execution is automatically accomplished by executing the PAM.BAT DOS batch file that was created by the MLLWDRIV code, as described earlier.

The program opens the input data files "PREDUST.DAT" and "RADUST.DAT" that were created previously and reads the values of the various variables. The input data file for the DUST code entitled "DUSTMLLW.INP" is then opened, and the following ten data sets required by the DUST code are written to the file:

Data Set 1: Title and General Problem Definition

Data Set 2: Time Parameters

Data Set 3: Material Assignments/Properties

Data Set 4: Output Specifications

Data Set 5: Facility Coordinate Data

Data Set 6: Initial and Boundary Conditions

Data Set 7: Water-Flow Parameters

Data Set 8: Container Parameters

Data Set 9: Waste Form Leaching Parameters

Data Set 10: External Sources

D-2.2 Verification

The PREDUST.FOR is a peripheral computer code that manipulates the data read in from the data files "PREDUST.DAT" and "RADUST.DAT" created by execution of the MLLWDRIV program. The program then creates the appropriate input data file, "DUSTMLLW.INP," for the DUST code for each radionuclide using data contained in the PREDUST.DAT and RADUST.DAT files, as well as values of some parameters that are hard-wired in the PREDUST code. The program also creates the input data file "POSTDUST.DAT" for the POSTDUST program.

Successful verification of the code was demonstrated by comparing the data in the input files and output files to ensure that the values were correctly transferred and in the proper format.

D-3. POSTDUST.FOR

D-3.1 Description

POSTDUST.FOR is a postprocessor program of the Model, written in FORTRAN, that converts the source-term data calculated by the DUST code into the form required by the WELLC and SURFQ codes. Each run of the DUST code writes the source-term data for a particular radionuclide XX to the file TRACEFXD.DAT. The POSTDUST code reads the flux (in units of Ci/cm²/yr) versus time (in yr) data from the TRACEFXD.DAT and multiplies each flux value by the cross-sectional area through which flow occurs within the facility. The waste form is considered impermeable relative to the backfill, and the cross-sectional area of flow corresponds to the area occupied by the backfill around the drums of waste. The resulting flux (in Ci/yr)-versus-time data are then written to the file DSTRADXX.DAT for the particular radionuclide XX.

D-3.2 Verification

Verification of the POSTDUST postprocessor was performed through a hand-calculation by multiplying the flux values in the TRACEFXD.DAT file by the cross-sectional flow area and then comparing the results to the DSTRADXX.DAT file generated by the POSTDUST program.

D-4. DECAYDOS.FOR

D-4.1 Description

The DECAYDOS.FOR is a postprocessor program of the Model that was written in FORTRAN that performs decay-chain calculations. The program reads the well-concentration-versus-time data in file WELLGEN.INP created by the WELLC code and accounts for the decay and in-growth of radionuclides. The resulting concentrations are multiplied by radionuclide-specific dose conversion factors to derive dose versus time profiles for each decay chain.

Based on screening of radionuclides in the alpha low-level waste (ALLW) inventory at the Idaho National Engineering Laboratory (INEL), eight radionuclides were considered for more detailed evaluations using the Model. The inventories of two of these radionuclides (Cm-244 and Pu-241) were combined with the inventories of their progeny (Pu-240 and Am-241, respectively), since the half-lives of Cm-244 and Pu-241 are short compared to the half-lives of their progeny.

The radioactive decay chains included in the current version of the Model are shown in Figure D-1. Each chain was considered to be a straight chain. (Any branching involved only very short-lived radionuclides, and the branches eventually rejoined at a common radionuclide.) Very short-lived radionuclides (for which dose data were not available in GENII) are not shown in Figure D-1 since this dose is included in the dose conversion factors for their parent radionuclides.

The decay and in-growth of radionuclides was performed within the DECAYDOS code using the following Bateman decay equations for straight-chain radioactive decay (Benedict et al. 1981):

$$N_i = N_1^0 \lambda_1 \lambda_2 \dots \lambda_{i-1} \sum_{j=1}^i \frac{e^{-\lambda_j t}}{\prod_{\substack{k=1 \\ k \neq j}}^i (\lambda_k - \lambda_j)} \quad (i > 1) \quad (1)$$

and

$$N_1 = N_1^0 e^{-\lambda_1 t} \quad (i = 1) \quad (2)$$

where

- N_i = number of atoms of radionuclide i
- N_1^0 = number of atoms of the progenitor radionuclide
- λ_i = decay constant for radionuclide i
- t = time.

Figure D-1. Radionuclide decay chains considered in the model and their corresponding half-lives.

The concentrations and fluxes of the progenitors, in terms of activity, were converted into equivalent atom concentrations and fluxes of the progenitors before applying the Bateman equations. The calculated concentrations and fluxes of the parents and progeny, in terms of atoms, were then converted into units of activity concentrations and fluxes for use with the Dose Integrated Over Ten Thousand Years code.

D-4.2 Verification

The DECAYDOS code was verified by comparing the predicted results with a code specifically developed for verification using a different calculational methodology. Specifically, the verification code DECHDOSE solves a system of differential equations representing a radioactive decay chain numerically. The procedure recommended for solving sets of stiff differential equations (Press et al. 1990) served as the core of this code. The codes were shown to provide identical results when the time-step size in the DECHDOSE program was set to less than one year.

REFERENCES FOR APPENDIX D

Benedict, M., T. H. Pigford, and H. W. Levi, 1981, *Nuclear Chemical Engineering*, 2nd ed., McGraw-Hill Book Company, New York, New York.

Press, W. H., B. P. Flannery, S. A. Teukolsky, and W. T. Vetterling, 1990, *Numerical Recipes. The Art of Scientific Computing FORTRAN Version*, Cambridge University Press, Cambridge, Massachusetts.

Appendix E

Description of Computer Programs

Appendix E

Description of Computer Programs

This appendix provides a brief description of the computer programs used, other than the preprocessor and postprocessor programs described in Appendix D.

E-1. DESCRIPTION OF VS2DT

The flow of groundwater through the disposal facility was modeled using the Variably Saturated 2-Dimensional Flow and Transport (VS2DT) code. The program uses a finite-difference approximation to the advection-dispersion equation with added retardation and decay to model both saturated and unsaturated flow and transport (Healy 1990). VS2DT is a nonproprietary computer code published by the U.S. Geological Survey (USGS) (Healy 1990).

The code is well documented and versatile and has been used to simulate variably saturated flow and solute transport under a variety of hydrogeologic conditions. Simulated regions can be one-dimensional columns, two-dimensional vertical cross sections, or axially symmetric, three-dimensional cylinders.

For this project, VS2DT was used to calculate the moisture content and Darcy velocity through the facility.

E-2. DESCRIPTION OF DUST

The Disposal Unit Source Term (DUST) code (Sullivan 1993) was selected to model release and transport of contaminants to the disposal facility boundary. The code accounts for the most important processes of fluid flow, container degradation, waste-form leaching, and radionuclide transport and decay which influence release.

DUST allows waste form release to be modeled in three ways: surface washoff, diffusion release, and uniform dissolution of the waste form. All of these release mechanisms account for radioactive decay of the source.

Two different methods are available for calculating transport of contaminants through the facility: the Multi-Cell Mixing Cascade (MCMC) model and the one-dimensional Finite Difference (FD) model. The FD model is much more flexible and is capable of simulating more processes than the MCMC model. The FD model checks to ensure that radionuclide solubilities are not exceeded, whereas the MCMC model does not perform this check. The FD model also permits the water velocity through the waste to be defined as a function of time. The MCMC model only allows a constant velocity to be defined. The MCMC model does not model diffusional releases, whereas the FD model does. Thus, the FD model was used for the performance assessment (PA) source-term.

The FD model solves the transport equation with the processes of advection, dispersion, retardation, and radioactive decay (Sullivan 1993). The FD model is one-dimensional and divides the model domain into control volumes. It then transforms the advection-dispersion transport equation into a set of coupled algebraic equations using the control volumes. Then a mass balance is performed on each control volume using finite differences.

For this project, DUST was used to provide a flux of radionuclides out the bottom of the disposal facility.

E-3. DESCRIPTION OF WELLC

The WELLC code consists of a vadose zone flow and transport routine and a saturated zone flow and transport routine. The code accepts as input the time-dependent flux of radionuclides entering the vadose zone, and returns as output the concentrations of radionuclides at a receptor well as a function of time.

The vadose zone routine considers one-dimensional advection and retardation in a variably saturated medium. Within the vadose zone, the average velocity and travel time of a nonsorbing contaminant moving from the waste zone to the water table is estimated analytically as a function of the infiltration rate, the degree of saturation, the porosity, and the thickness of the vadose zone. The travel time of each contaminant is then calculated by multiplying the vadose zone travel time by the retardation coefficient for that contaminant.

Within the saturated zone, calculations are performed using Green's function solutions to the steady, one-dimensional aquifer flow, advection-dispersion equations with added retardation. To solve the Green's function, the code requires properties of the aquifer, disposal facility, and the location of a receptor well. In addition, the mass flux into the aquifer from the vadose zone flow and transport routine is used as the source of contamination to the aquifer. The aquifer properties required are retardation coefficients, thickness, groundwater velocity, and longitudinal and transverse dispersivity. The disposal facility properties required are the length of the facility along the direction of groundwater flow, the width of the facility perpendicular to the groundwater flow direction, and the distance to a receptor well from the downgradient edge of the facility. This solution is then used over various times to develop a concentration-versus-time profile at the receptor well location.

E-4. DESCRIPTION OF GENII

The Generation-II (GENII) package (Napier et al. 1988) was used to calculate the doses for the intruder scenarios. GENII is one of the most commonly used packages in Department of Energy (DOE) low-level waste (LLW) disposal facility PAs. It implements the recommended U.S. Nuclear Regulatory Commission (NRC) exposure pathways models and performs dose calculations using the methods recommended by the International Committee on Radiation Protection.

The GENII package has the capability to calculate radiation dose to individuals and populations from both acute and chronic releases from all contributing pathways of exposure, including external exposure and ingestion. The GENII package comprises seven interrelated computer codes and their associated data libraries. The codes fall into three categories: user interface, internal and external dose factor generators (codes INTDF and EXTDF), and environmental dosimetry programs. The EXTDF module allows easy and uniform calculation of external dose-rate factors for submersion in an infinite cloud of radioactive materials, immersion in contaminated water, and direct exposure to plane or slab sources of contamination. The EXTDF code is a modification and enhancement of the well-known shielding code ISOSHL. The INTDF code is used to make estimates of the dose equivalents in a number of target organs from the activity in a given source organ. Most of the INTDF code is used to calculate the integrated retentions of parent and progeny radionuclides in the source organs. The

environmental dosimetry programs can be used to evaluate either short- or long-term scenarios. The short-term environmental dosimetry portion is divided into three separate but interrelated programs (ENVIN, ENV, and DOSE) that handle input organization and checking, environmental exposure, and dose calculations, respectively. The ENVIN module controls the reading of the input files. The ENVIN code interprets the basic input, reads the basic GENII data libraries and other optional input files, and organizes the input into sequential segments based on radionuclide decay chains. The ENV module calculates the environmental transfer, uptake, and human exposure to radionuclides that result from the chosen scenario for the defined source term. The DOSE module reads the annual intake and exposure rates defined by the ENV module and converts them to radiation dose. The calculation of external dose is done with precalculated factors from the EXTDF module, and the calculation of internal dose is done with precalculated factors from the INTDF module. This portion of the GENII package is designed to analyze environmental consequences in the near term (0 to 100 yr) following release to the environment and was applied to evaluate the impacts of the three human-intrusion scenarios. The primary limitation of the short-term portion is that the programs are based on steady-state radionuclide concentrations in environmental media.

REFERENCES FOR APPENDIX E

- Healy, R. W., 1990, "Simulation of Solute Transport in Variably Saturated Porous Media with Supplemental Information on Modifications to the U.S. Geological Survey's Computer Program VS2D," *Water-Resources Investigations Report 90-4025*, U.S. Geological Survey, Denver, Colorado.
- Napier, B. A., R. A. Peloquin, D. L. Streng, and J. V. Ramsdell, 1988, *GENII-The Hanford Environmental Radiation Dosimetry Software System: Conceptual Representation*, PNL-6584, Vol. 1, Pacific Northwest Laboratory.
- Sullivan, T. M., 1993, *DUST Disposal Unit Source Term: Data Input Guide*, NUREG/CR-6041, BNL-NUREG-52375, Brookhaven National Laboratory, April.

Appendix F

Verification and Validation of Computer Programs

Appendix F

Verification and Validation of Computer Programs

This appendix discusses the verifications and validations of the computer programs described in Appendix E. The process of verifying, validating, and testing software was considered essential in the development of the Model. Verification is an independent assessment of the numerical correctness of the software results. Validation is an independent assessment of the theoretical correctness of the software.

The Model is made up of Nuclear Regulatory Commission (NRC) computer codes and computer codes widely used in the Department of Energy (DOE) system, as well as the pre- and postprocessors that have been developed to link the individual codes into the Model. The VS2DT, DUST, and GENII codes are considered "industry standards," and verification and validation (V&V) was performed outside the current effort. The pre- and postprocessors MLLWDRIV, PREDUST, and POSTDUST, which simply manipulate input and output files required and created by the industry standard codes, were verified by confirming that the required data and formats were properly transferred between files (see Appendix D). A number of calculations are performed within the DECAYDOS postprocessor of the Model. This program was verified/validated by comparing the results to those of an independently developed program that uses a different calculational methodology. The following describes the verification and validation that have been performed for the various codes.

F-1. V&V OF VS2DT

The flow through the disposal facility was modeled by using the VS2DT model. VS2DT is a U.S. Geological Survey (USGS) finite difference code that can be used to model both saturated and unsaturated flow and transport (Healy 1990). Recent discussions with the Sandia National Laboratories/New Mexico (SNL/NM) staff, who are developing the NRC performance assessment (PA) methodology, indicated that VS2DT will be recommended as an alternative model to the previously recommended VAM2D model (Kozak and Olague 1993). VS2DT is a nonproprietary computer code published by the USGS. The code is well documented and versatile and has been used to simulate variably saturated flow and solute transport under a variety of hydrogeologic conditions.

A study has recently been completed to identify the "code of choice" for the SNL/NM Environmental Restoration Program to use to simulate vadose zone and contaminant transport. As part of the study, SNL/NM staff benchmarked VS2DT against VAM2D (Huyakorn et al. 1989), a proprietary finite-element flow and solute transport code that is widely used in DOE site PAs. The SNL/NM staff found that the codes gave comparable results but that VS2DT was significantly faster (McCord and Goodrich 1993).

The VS2DT code was developed by the USGS under their software quality assurance (QA) standards. It had been tested and verified previously against several analytical solutions (Healy 1990). Although several verification examples were provided in the VS2DT manual (Healy 1990),

it was felt that some additional, independent verification was in order before an unequivocal recommendation to use the VS2DT code on this project could be given. Two verification problems were selected based on two analytical solutions. The first verification problem represented two-dimensional unsaturated flow to a water table that was solved analytically by Philip (1989). The second verification problem simulated two-dimensional saturated solute transport and was solved analytically by Javandel et al. (1984) and named "TDAST." The differences in mass between that predicted by the VS2DT code and the prediction in the analytical solution were less than 2% for each verification problem.

F-2. V&V OF DUST

Verification and validation of the DUST code have been conducted through a number of studies. The container degradation models (general and local failure) and the waste form release models (diffusion, uniform, and surface rinse with partitioning) were independently verified by comparing the predicted results of known analytical solutions. The finite difference and multi-cell mixing cascade models were extensively tested by comparing the predicted results to known analytical solutions and by comparing the results predicted by the two models on identical problems. In all cases, there was excellent agreement among the various results.

F-3. V&V OF WELLC AND SURFQ

Because the source codes for DISPERSE and SURFACE that were developed and recommended by Kozak et al. (1990) were unavailable, comparable codes were developed for use in the Model. The WELLC and SURFQ codes of the Model provide the concentrations of radionuclides at the well and the flux of radionuclides entering a surface-water body, respectively, as a function of time. In both codes, the calculations are performed using Green's function solutions to the steady, one-dimensional aquifer flow, advection-dispersion equations. The WELLC and SURFQ codes thus use the same equations as in the corresponding DISPERSE and SURFACE codes, respectively, and differ only in the numerical integration scheme. Specifically, in the WELLC and SURFQ codes, integration of the product of the Green's function and the source rate is performed using the Quadrature routine Adaptive integrator General purpose user-defined integer and Extended parameters list for increased flexibility (QAGE) routine of QUADPACK (Piessens et al. 1983).

The codes were tested and successfully verified through benchmarking exercises with the DISPERSE and SURFACE codes. Each subroutine of the DISPERSE and SURFACE codes was verified independently before being used in the code. The codes were then verified for self-consistency by comparing results for a point source to results from the point-source of Green's function. The codes were also verified against the analytical Green's function for an area source of infinitesimal duration, and the codes reproduced the analytical results well. The results of the DISPERSE code were compared to those predicted by the FEMWATER and the VAM2D codes for a simple conceptual model. The effects of several parameters on the comparisons were examined for each case. The results from the three codes are qualitatively similar, but the simpler DISPERSE code produced slightly more conservative well concentrations.

F-4. V&V OF GENII

The GENII package was developed in accordance with American Society of Mechanical Engineers standard NQA-1 as implemented in the Pacific Northwest Laboratories Quality Assurance Manual. The design process consisted of developing and internally testing software, developing test cases, and documenting software in accordance with the design input. The GENII package has been extensively tested and verified by hand, using hand-calculation worksheets, and has been benchmarked against similar Hanford environmental dosimetry programs. The code has also undergone extensive peer reviews. Additional GENII QA-related information is provided in Chapter 5.0 of the GENII Users' Manual (Napier et al. 1988).

REFERENCES FOR APPENDIX F

- Healy, R. W., 1990, "Simulation of Solute Transport in Variably Saturated Porous Media with Supplemental Information on Modifications to the U.S. Geological Survey's Computer Program VS2D," *Water-Resources Investigations Report 90-4025*, U.S. Geological Survey, Denver, Colorado.
- Huyakorn, P. S., J. B. Kool, and J. B. Robertson, 1989, *Documentation and User's Guide: VAM2D—Variably Saturated Analysis Model in Two Dimensions (Version 5.0 with Hysteresis and Decay Transport)*, NUREG/CR-5352, HCL/89-01, HydroGeoLogic Inc., Herndon, Virginia.
- Javandel, I., C. Doughty, and C. Tsang, 1984, "Groundwater Transport: Handbook of Mathematical Models," *Water Resources Monograph Series 10*, American Geophysical Union, Washington, D.C.
- Kozak, M. W., and N. E. Olague, 1993, "Updated Recommendations for Low-Level Waste Performance Assessment," *Proceedings of the Symposium on Waste Management*, Tucson, Arizona, March, pp. 371–376.
- Kozak, M. W., M. S. Y. Chu, and P. A. Mattingly, 1990, *A Performance Assessment Methodology for Low-Level Waste Facilities*, NUREG/CR-5532, SAND90-0375, Sandia National Laboratories, Albuquerque, New Mexico.
- McCord, J. T., and M. T. Goodrich, 1993, *Benchmark Testing and Independent Verification of the VS2DT Computer Code*, SAND91-1526, draft, Sandia National Laboratories, Albuquerque, New Mexico.
- Napier, B. A., R. A. Peloquin, D. L. Streng, and J. V. Ramsdell, 1988, *GENII—The Hanford Environmental Radiation Dosimetry Software System: Conceptual Representation*, PNL-6584, Vol. 1, Pacific Northwest Laboratory, Richland, Washington.
- Philip, J. R., 1989, "Multidimensional Steady Infiltration to a Water Table," *Water Resources Research*, Vol. 25, No. 1, pp. 109–116.
- Piessens, R., E. deDoncker-Kapenga, C. W. Uberhuber, and D. K. Kahener, 1983, "QUADPACK—A Subroutine Package for Automatic Integration," *Springer Series in Computational Mathematics*, Vol. 1. Springer-Verlag, Berlin, Germany.

Appendix G

Final Waste Form Characteristics

Appendix G

Final Waste Form Characteristics

This appendix provides the properties of untreated waste and final waste forms (FWFs) that were used.

Three categories of FWFs—cement, glass, and ceramic—were selected for evaluation. For each category, two FWFs were identified to represent realistic limits of the behavior of relatively favorable FWFs and relatively unfavorable FWFs. As used here, the term "favorable" refers to FWF behavior that tends to enhance long-term immobilization of the radionuclides.

For some properties, measured values for specific FWFs were used. Those specific FWFs are as follows: Portland cement concrete (PCC) and concrete fused under elevated temperature and pressure (FUETAP); borosilicate glass (BSG) and aluminosilicate glass (ASG); and iron-enriched basalt (IEB) and IEB with TiO_2 and ZrO_2 additions (IEB4). For other properties, the ranges of values represent variations that may occur in practice during treatment operations, rather than correlations with a particular FWF. For such properties, the analogy was continued by assigning the more favorable value of the property to the more favorable FWF, and the less favorable value to the less favorable FWF. For simplicity, the FWF terminology used in the property tables is "More Favorable" and "Less Favorable," for the ranges of FWFs under each of the three categories.

Table G-1 lists the values used for the FWF properties that are independent of contaminant. The specific gravity and porosity of untreated waste are also addressed. Table G-2 lists the values used for the FWF properties that are contaminant-dependent. The contaminants for which the values apply are indicated. The values listed in Tables G-1 and G-2 were compiled from various IT Corporation and Idaho National Engineering Laboratory (INEL) reports. Table G-3 lists typical chemical compositions of the FWFs.

Also included in this appendix are two letters that provide information on FWF characteristics. The first letter, from T. L. Eddy on January 18, 1995, is titled "Leach Rate Updates Based on Recent Experiments." It provides updated values for the waste form dissolution rates in Table G-1, based on data published after this performance assessment (PA) task had begun. The data from the Eddy letter that were used in the PA runs are those listed under the heading "New MCC-1" in Table 6. These data are referred to in the main body of this report as the "revised leach rates."

The second letter, from T. L. Eddy on March 14, 1995, is titled "Two-Step Leach-Rate Decay Model." It provides guidance rates in different time periods. The data that were used in the PA runs appear in Table II of the letter. Runs were made only for the IEB FWF. The step 1 value of $0.08 \text{ g/m}^2\text{-day}$ was used. For conservatism, the step 2 value used was the largest for the BSG FWF. These data are referred to in the main body of this report as the "time-dependent leach rates."

Table G-1. Final waste form properties that are independent of contaminant.

Property	Untreated waste	Less favorable cement	More favorable cement	Less favorable glass	More favorable glass	Less favorable ceramic	More favorable ceramic
Example of FWF type ^a	NA ^b	PCC	FUETAP	BSG	ASG	IEB	IEB4
Chemical composition of waste form matrix	NA	See Table G-3					
Shape and surface area per unit volume	NA	[Assume all FWFs are in 55-gallon drums]					
Waste form dissolution rate (g/m ² -day)	NA	NA	NA	0.4	0.07	0.08	0.05
Specific gravity	0.65	1.95	1.95	2.65	2.65	3.0	3.5
Porosity	NA	22%	7%	NA	NA	NA	NA

a. The abbreviated designations represent the following FWFs:

PCC = Portland cement/concrete
 FUETAP = formed under elevated temperature and pressure (cement)
 BSG = borosilicate glass
 ASG = aluminosilicate glass
 IEB = iron-enriched basalt
 IEB4 = enhanced iron-enriched basalt

b. NA = Not applicable.

Table G-2. Final waste form properties that are contaminant-dependent.

Property	Less favorable cement	More favorable cement	Less favorable glass	More favorable glass	Less favorable ceramic	More favorable ceramic
Concentration of individual radionuclides in FWF (Ci/cm ³)	[See Appendix A, Table A-7]					
Availability	[100% for all contaminants]					30% for U, Pu, Am, Cm, Sr; 100% for all other contaminants
Effective diffusion coefficient (cm ² /sec)	NA					
Sr	4.12 x 10 ⁻⁸	1.18 x 10 ⁻⁸				
Y	2.96 x 10 ⁻⁸	8.44 x 10 ⁻⁹				
Ba	1.02 x 10 ⁻⁷	2.90 x 10 ⁻⁸				
Co	3.55 x 10 ⁻⁸	1.01 x 10 ⁻⁸				
Cs	1.10 x 10 ⁻⁷	3.13 x 10 ⁻⁸				
U	2.52 x 10 ⁻⁸	7.19 x 10 ⁻⁹				
Pu	3.08 x 10 ⁻⁸	8.79 x 10 ⁻⁹				
Am	3.07 x 10 ⁻⁸	8.77 x 10 ⁻⁹				
Cm	2.98 x 10 ⁻⁸	8.53 x 10 ⁻⁹				

Table G-2. (continued).

Property	Less favorable cement	More favorable cement	Less favorable glass	More favorable glass	Less favorable ceramic	More favorable ceramic
Solubility limits (g/cm ³)						
Am	4.60×10^{-7}					
Pu	4.51×10^{-8}					
U	3.03×10^{-5}					
Cm	3.03×10^{-5} (conservative value)					
All other elements	1×10^{-2} , so as not to limit releases					

Table G-3. Typical chemical composition of final waste forms (wt%).

FWF	SiO ₂	B ₂ O ₃	Na ₂ O	K ₂ O	Li ₂ O	MgO	CaO	FeO & Fe ₂ O ₃	Al ₂ O ₃	TiO ₂	ZrO ₂	H ₂ O	Misc
PCC ^a	25.2		1.9	1.0		1.8	15.3	14.0	5.4			35.0	
FUETAP ^b	45.7		1.9			1.8	14.5	19.7	5.2	3.8		2.0	4.5
BSG ^c	49.6	7.0	9.6		4.9	0.7	1.1	15.8	4.5		0.7		6.0
ASG ^d	53.2		1.1	1.9		2.8	12.0	19.0	9.7	0.4			
IEB ^e	55.7		2.6	2.7		3.1	10.3	13.8	11.4	0.4			
IEB4 ^f	51.7		2.4	2.5		2.9	9.6	12.8	10.6	5.0	2.5		

a. Grandy, J. D., et al., 1993, p. 10. Assumes 40% A-Series waste loading.

b. Dole, L. R., et al., 1983, p. 5-6. Assumes 40% A-Series waste loading, 18.7% Portland cement, 9.4% flyash, 6.4% clay, 23.6% sand, and 2% water.

c. Kong, P. C., et al., 1993, Table 12. BSG with 30% waste loading. (Document is in draft form and not yet available for public distribution.)

d. Grandy, J. D., et al., 1993, Table p. A-5, S-60 mixture.

e. Grandy, J. D., et al., 1993, A-60 mixture.

f. Grandy, J. D., et al., 1993, A-60 mixture prorated for 7.5% TiO₂ + ZrO₂.

REFERENCES FOR APPENDIX G

- Dole, L. R., G. C. Rogers, M. T. Morgan, D. P. Stintion, J. H. Kessler, S. M. Robinson, and J. G. Moore, 1983, *Cement-Based Radioactive Waste Hosts Formed Under Elevated Temperatures and Pressures (FUETAP Concretes) for Savannah River Plant High-Level Defense Waste*, ORNL/TM-8579, Oak Ridge National Laboratory, April.
- Grandy, J. D., T. L. Eddy, and G. L. Anderson, 1993, *TSA Waste Stream and Final Waste Form Composition*, EGG-MS-10617, January.
- Kong, P. C., G. A. Reimann, S. O. Bates, B. A. Detering, T. L. Eddy, and G. L. Anderson, 1993, *Final Waste Form Option Scoping Studies for RWMC Wastes*, EG&G Idaho, Inc., EGG-WTD-10363, Draft, August.

INTERDEPARTMENTAL COMMUNICATION

Date: January 18, 1995
To: T. H. Smith, MS 3960, FAX 6-3612
From: T. L. Eddy, MS 2210
Subject: LEACH RATE UPDATES BASED ON RECENT EXPERIMENTS

Leach rates previously provided for the performance assessment were obtained from a survey of existing 28-day MCC-1 leach rate data, predominantly via *Radioactive Waste Forms of the Future* by W. Lutze and R. C. Ewing¹, North-Holland, 1988 and as tabulated by Eddy². The exception being bulk leach rates and availability factors for IEB4, the more favorable ceramic.

Recent leach rate measurements obtained on the PSPI Proof Test project are now available to check the values used previously. Leach tests were made on the less-favorable and more-favorable ceramic, IEB and IEB4, respectively. Attempts to measure the leach rates of aluminasilicate glass (ASG) were unsuccessful, as such, because the ASG turned rapidly into a glass-ceramic, ASGC, even with the "fast-cooled" procedure.

The evaluation is more complicated than originally expected due to the following facts:

1. Bulk rates are desired, but were not measures directly. Usually the Si leach rate is close to the effective bulk leach rate and can be substituted.
2. The leach rates for the radionuclides are usually much (one to four orders of magnitude smaller) than the bulk or Si rate. Sometimes the "availability factor" is used to account for this large difference.
3. MCC-1 rates were obtained only for certain samples. PCT rates were obtained for all samples. A conversion method has been derived, which has relatively good accuracy when fitting over several orders of magnitude, but possibly not when at the higher values of from 0.1 to 1.0 g/m²-d.

¹W. Lutze and R. C. Ewing, *Radioactive Waste Forms of the Future*, North-Holland, 1988.

²T. L. Eddy, "Selection of a Final Waste Form," draft paper, March 1993 and Ch. 5 of the still unpublished *Final Waste Form Option Scoping Studies for RWMC Wastes*, draft report, August 1993.

4. Values were obtained for different compositions: alumina-silicate glass-ceramic (ASGC), IEB as A-40, and IEB4 as A-40 and H1-0 to H1-100.
5. Some of the glass which was fast-cooled (FC), became devitrified, resulting in two devitrified samples: fast-cooled and devitrified (12 h at 1200°C) samples. Crystalline ceramic samples were slow-cooled (SC) or devitrified (DV).
6. MCC-1 values in $\text{g/m}^2\text{-d}$ are grams leached per unit surface area of the monolith divided by the concentration of the leached oxide and the number of days. PCT values in $\text{g/m}^2\text{-d}$ are grams leached per assumed surface area (2000 or $1955 \text{ m}^2/\text{m}^3 = S/V$) divided by the concentration of the leached oxide and the number of days. These are not equivalent because the effective leaching area is not known in the PCT data and the average rate will be different over the different number of days (7 vs. 28).

The values obtained from the recent PSPI MCC-1 leach test measurements are in Table 1. The last column contains values gleaned from work at the INEL (using a more complex devitrification cycle) as summarized in Lutze & Ewing.¹ The new PSPI MCC-1 measurements are at the high end of the range given in the last column.

Table 1. MCC-1 leach rate values in $\text{g/m}^2\text{-d}$.

	ASGC S-40 (FC)	ASGC S-40 (DV)	IEB A-40 (DV)	IEB4 A-40 (DV)	IEB4 H1-40 (DV)	IEB & IEB4 (DV)
Si/Bulk	0.13	0.21	0.10	0.11	0.27	0.01-0.14
Ca	0.077	0.14	0.077	0.11	0.14	0.03-0.13
Na	1.7	3.4	0.50	0.56	0.96	0.05
Cs	0.78	0.73	0.44	0.28	0.74	0.06-0.74

Table 2 contains the PSPI and WSL PCT measurements with measured values normalized by the leached species concentrations based on SEM/EDXS, but without other corrections or conversions.

Table 2. PCT-Method A 7-day leach rates in normalized mg/L .

Species	H1-0	H1-20	H1-40	H1-60	H1-80	H1-100	EA Glass
Si SC	2.16E+01	8.34E+02	2.35E+02	1.74E+02	9.08E+01	6.95E+01	3.92E+03
DV	1.92E+03	1.18E+03	2.76E+02	1.95E+02	1.41E+02	7.89E+01	
Ca SC	3.89E+00	6.67E+00	8.53E+00	4.34E+00	1.63E+01	5.93E+01	
DV	<1.73E-02	3.74E+00	4.01E+00	3.43E+00	6.51E+00	5.91E+01	
Na SC	2.87E+04	5.55E+01	2.50E+03	1.63E+03	3.78E+02	4.59E+02	1.33E+04
DV	2.21E+04	8.34E+03	7.54E+01	1.63E+03	1.03E+03	4.98E+02	
Cs SC	7.56E+04	4.50E+02	9.54E+02	5.49E+02	2.06E+02	6.07E+01	
DV	5.49E+04	1.95E+03	1.40E+03	5.80E+02	2.95E+02	9.26E+01	
Ce SC	<2.04E-02	1.00E+00	2.05E-01	1.62E+00	1.22E-01	6.05E-01	
DV	<4.76E-02	1.28E-01	7.59E+00	3.69E+00	9.56E-01	2.02E-01	
Sm SC	<5.66E-02	1.43E+00	1.92E-01	4.76E-01	1.43E-01	4.43E-01	
DV	<6.90E-02	1.47E-01	1.06E+01	5.34E+00	9.82E-01	2.08E-01	

Table 3 contains PCT values in "MCC-1" units as calculated by Rick Farnsworth. I have been unable to determine how these values were calculated. The normalization used the wet chemical analysis compositions.

Table 3. PCT values in g/m²-d after Farnsworth.

	ASGC	ASGC	IEB	IEB4	IEB4
	S-40	S-40	A-40	A-40	H1-40
	(FC)	(DV)	(DV)	(DV)	(DV)
Si/Bulk	0.0071	0.0070	0.0096	0.0091	0.020
Ca	0.00075	0.0026	0.00051	0.00079	0.00037
Na	0.12	0.051	0.070	0.062	0.0056
Cs	0.032	0.012	0.035	0.011	0.045

Table 4 contains the PCT measured values from Table 2 and 4 converted to MCC-1 equivalent values from the first, rough, manual scheme developed by Eddy & Grandy.^{3,4} The significance of this table is that the values are based on a surface area to volume ratio (S/V) of about 500:1, in contrast to the 2000:1 prescribed by the developers of the PCT method. Second, the values in Table 4 agree very well with Farnsworth's values in Table 3.

Table 4. PCT values (g/m²-d) converted to MCC-1 manually with S/V ~ 500.

	ASGC	ASGC	IEB	IEB	IEB4	IEB4	IEB4	IEB4
	S-40	S-40	A-40	A-40	A-40	A-40	H1-40	H1-40
	(FC)	(DV)	(SC)	(DV)	(SC)	(DV)	(SC)	(DV)
Si/Bulk	0.0073	0.0069	0.0099	0.0096	0.013	0.0099	0.018	0.021
Ca	0.00056	0.0021	0.00027	0.00043	0.00024	0.00069	0.00066	0.00031
Na	0.28	0.092	0.094	0.064	0.15	0.081	0.19	0.0058
Cs	0.053	0.070	0.0056 ⁵	0.00077	0.0027	0.023	0.073	0.11

Comparing the values of Tables 3 & 4 with those in Table 1, one sees that the PCT values are much smaller, even though the PCT surface area is almost 2000 times larger and the average leach rate should be many times larger over the first 7 days than over 28 days.

This can be resolved by recognizing that the effective surface area is much less than 2000 or even 500 times the sample volume because of contacts between

³T.L. Eddy & J.D. Grandy, Interoffice correspondence TLE-3-94, Feb. 2, 1994.

⁴T. L. Eddy, et al., "Properties of Vitrified Rocky Flats TRUW with Different Waste Loadings," *Ceramic Transactions* 45, Am. Cer. Soc., 1994.

⁵Cs wt% not measured. Assumed to be 0.50 wt% as typical of other tests.

particles and reduced diffusion rates. Eddy, Grandy, Trimberger and Anderson⁶ found a S/V of approximately 25 instead of 2000, as determined with a statistical best curve fit of PSPI, WSL and previous data including lanthanides (Ce & Sm). The values of Tables 2 and 4 with S/V = 25 are given in Table 5. The converted-PCT values in Table 5 now are in good agreement with the measured MCC-1 values in Table 1.

Table 5. PCT values in g/m²-d as converted to MCC-1 with S/V = 25.

	ASGC	ASGC	IEB	IEB	IEB4	IEB4	IEB4	IEB4
	S-40	S-40	A-40	A-40	A-40	A-40	H1-40	H1-40
	(FC)	(DV)	(SC)	(DV)	(SC)	(DV)	(SC)	(DV)
Si/Bulk	0.15	0.14	0.20	0.19	0.26	0.20	0.36	0.42
Ca	0.011	0.042	0.0054	0.0086	0.0048	0.014	0.013	0.0062
Na	5.6	1.8	1.9	1.3	3.0	1.6	3.8	0.116
Cs	1.1	1.40	0.11 ⁷	0.015	0.054	0.46	1.46	2.16

In a larger set of data for IEB4, H-0, H-20, H-40, H-60, H-80, and H-100 relatively consistent PCT values were obtained in which if normalizing on Na as 1.0, then Cs is 0.4, Si is 0.1, Ca is 0.002, and the transuranic surrogates are 0.0003. These values are in agreement with the values in Table 5 and Table 1.

Conclusions

A comparison of prior and new measurement values is given in Table 6 based on the Si leach rate as the bulk rate. ASG(original), ASGC(FC), IEB(DV) and IEB4/A-40(DV) with the S/V = 25 are used in the table.

Table 6: New Measurements and Prior Bulk Leach Rate Choices.

		Prior Choice	New MCC-1	New PCT	Availability Prior	Recom.
Least Favorable Glass	BSG	0.4	---	---	100%	100%
Most Favorable Glass	ASG/ASGC	0.07	0.13	0.15	100%	100%
Least Favorable Ceramic	IEB	0.08	0.10	0.19	100%	100%
Most Favorable Ceramic	IEB4	0.05	0.11	0.20	30%	30% ⁸

The new leach rate measurements, both direct MCC-1 and "corrected" PCT (to match the MCC-1) are a factor of two larger than those from prior, other sources. Whether this is due to the limited sample, the different waste form formation process, or differences in the analytical methods is not known. The

⁶T.L. Eddy, J.D. Grandy, B.R. Trimberger and G.L. Anderson, "Method of Comparison of PCT and MCC-1 Leach Rates," I&EC Special Symposium, American Chemical Society Meeting, Atlanta GA, Sept. 19-21, 1994.

⁷Cs wt% not measured. Assumed to be 0.50 wt% as typical of other tests.

⁸30% on U, Pu, Am, Cm, Sr. All other contaminants 100%.

trend in the new PCT values is identical to the prior choice, and would be nearly numerically equal if S/V = 50 or 60, instead of 25.

The IEB4/H1-40(DV) has a higher Si(bulk) leach rate than IEB4/A-40(DV) (MCC-1: 0.27 vs. 0.11 or PCT: 0.42 vs 0.20), but that may be in part due the lower concentration of SiO₂, 17.29 wt% vs. 21.58 wt% affecting normalization, and the much higher Na concentration, 4.78 wt% vs. 1.15 wt%. Leach rates in the H1-series increased linearly with Na content and decreased with Si content. When comparing the relative amounts of each of the major waste streams, the H1-series is relatively small and has the most Na. Most of the waste forms generated from the other waste streams will be more stable than the A-series waste, thus the A-series waste leach rate is best to use as a mean.

Recommendations

The recommendation is to use the same dissolution rates and availability factors as before and shown in Table 6 based on the above arguments with the following exceptions:

One recommended change is for IEB4, in which the bulk material is essentially the same as the IEB (except for less than 10 wt% of TiO₂ plus ZrO₂) and therefore should be adjusted to equal the IEB leach rate. This equivalence of IEB and IEB4 leach rates is borne out in the measurements and focussed upon in Table 5.

At the same time the IEB4 leach rate data (not included above because no MCC-1 measurements were reported) shows that the transuranic surrogates are even more unavailable than perceived from prior data and the availability factor could be lowered for U, Pu, Am, Cm, and Sr to 10% with good cause. This recommendation for Sr is supported above with the results for Ca, which is a Sr surrogate.

Distribution:

G. L. Anderson	MS 2414
P. C. Kong	MS 2210
O. L. Wiersholm	MS 3649
B. R. Raivo	MS 3765
N. R. Soelberg	MS 3625
S. M. Sorensen	MS 2210
S. O. Bates	MS 3710

MeltTran Corporation

2300 NORTH YELLOWSTONE HWY.
IDAHO FALLS, ID 83401
PHONE: (208) 524-6358; FAX: (208) 523-1049

March 14, 1995

Thomas H. Smith
INEL/LITCO, P.O. Box 1625
Idaho Falls, ID 83415-3960

Dear Tom:

Please find attached a copy of the letter report on the "Two-Step Leach-Rate Decay Model" which I was able to complete at MeltTran.

The results indicate that the originally proposed MCC-1 leach rate values (and availability factors) can be used for the 1st step of 1-year with excellent accuracy. The 2nd step leach rates are obtained using similarity principles from long term BSG leach rates proposed by Grambow (1988). Availability factors should be usable for this period, as well.

The initial approach that I was going to try did not work out. The subsequent approach resulted in a relatively simple method to extrapolate to long time periods. The accuracy of the method for long term rates depends on the accuracy of the BSG long time value for which there is no explicit experimental information to my knowledge. Test results closest to the desired measurements are being made at Argonne National Lab by Terry Johnson and Dave Wronkiewicz. These are the steam vapor tests at elevated temperatures. The results are being compared to natural analogues.

I hope the results are appropriate for your needs. If you have any questions, please contact me at MeltTran.

Sincerely yours,

Thomas L. Eddy

President

Enclosure

CC: Gary L. Anderson, LITCO

TWO-STEP LEACH-RATE DECAY MODEL

T. L. Eddy
MeltTran Corporation
2300 North Yellowstone HWY., Idaho Falls, ID 83401

March 10, 1995

Purpose

The purpose of this report is to propose a two-step model for leach rates to be used in performance assessment calculations, as requested by T. H. Smith, LITCO. This work was initiated at LITCO and completed at MeltTran. The first step would be over short time periods and may be based on the MCC-1 or equivalent leach rates. The second step would represent rates over many decades or centuries in the repository.

This work does not include the effect of pH, temperature, or other sophistications of leach behavior. The leach rates presented are based on deionized water at 90°C and are "normalized" to MCC-1 leach rates as previously supplied to IT Corp. for the PA.

Most of the referenced work is obtained from the monograph by Lutze & Ewing.¹

Glass Waste Forms

Figure 1 shows the cumulative fractional release of different species from BSG over time for two different ground-water flow rates as calculated by Grambow, et al.² With high rates of flow (100 L/y, dashed lines), the rates are internal-diffusion limited and form "nice and parallel" curves that represent initial release rates (at 10^{-2} = 3.65 days), logarithmic long-term rates (> 1 year), and the transition region between them. At low flow rates (1 L/y), the release becomes flow limited over a longer transition period up to 10 to 100 years. Also the fractional release may be "irregular" for some constituents, e.g., Al here.

The curves in Fig. 1 for glass waste forms could be approximated by a two-step leach process, approximated as shown in Fig. 2, with a constant fractional release rate to the knee of the curve followed by a logarithmic release rate over long time periods. In Figs. 1 & 2, the fractional amount released is the total amount leached divided by the initial amount in the glass; hence a cumulative total in per cent. (Note that different charts give different types of data.)

The performance assessment model will theoretically account for the driving leach-rate mechanisms: internal diffusion, external flow rate, etc.; therefore, the necessary dissolution rate is that for high flow rates which is internal-diffusion limited. Different leach conditions that are the responsibility of the

¹W. Lutze and R. C. Ewing (Eds.), *Radioactive Waste Forms for the Future*, North-Holland, Amsterdam, 1988

²B. Grambow, W. Lutze, R.C. Ewing, and L.O. Werme, in *Scientific Basis for Nuclear Waste Management XI*, Vol. 112, p. 531, M.J. Apted & R.E. Westerman (eds.), Materials Research Society, Pittsburgh, PA, 1988.

PA model are illustrated in Figure 3. They are not considered here.

Ceramic Waste Forms

Though Fig. 1 is for BSG, the characteristics should be valid for the glass phase of similar glass ceramics as considered here. It may not be valid for the crystalline ceramic phases for reasons illustrated in Fig. 4 from Reeve et al.³ The figure illustrates that the BSG glass instantaneous leach rate is flat over the significant time period of 75 days; whereas, in the crystalline/ceramic Synroc-C, the leach rate of the same substances is (Cs & Sr) decreases significantly at a semilogarithmic rate. Other differential curves appear to decrease exponentially as shown in Figure 5.⁴ Both are with deionized water that is replaced daily and the shape does not appear to be affected significantly by temperature (Fig. 5).

A third representation is the cumulative mass flux (species) leached versus time as shown in the semilogarithmic plot of Figure 6.⁵ As suggested in the differential or instantaneous leach plots, Fig. 6 shows a trend toward linearity at long times (in days). These three types of plots help one to understand the different mechanisms at play and how to model them for the present work.

Derivation of the Two-Step Model

Since the fractional release rates are constant for about one year, they should be calculable from the MCC-1 or converted PCT values with reasonable accuracy. The latter are normalized on initial concentrations, but are usually at very short time periods compared to one year; therefore, the initial concentrations should be good approximations.

The MCC-1 normalized mass flux leach rate L_{nmf} is specified in g/m²-d which is actually:

$$\frac{(\text{grams of substance leached})}{(\text{grams of substance in waste})} \frac{(\text{grams of waste})}{(\text{surface area in m}^2)} \frac{1}{(\text{no. of days})}$$

This can be converted to a fractional release rate:

$$\frac{(\text{grams of substance leached})}{(\text{grams of substance in waste})} \frac{1}{(\text{surface area in m}^2)} \frac{1}{(\text{no. of days})}$$

by dividing by total mass of the waste. Since all the "samples" have the same mass and since we really want L_{nmf} and not L_{nmf}/m_{waste} , the two variables are related

³K.D. Reeve, D.M. Levins, E.J. Ramm, and J.L. Woolfrey, "The Development and Evaluation of SYNROC for High Level Waste Immobilization," *Proc. IAEA/CEC/NEA Int. Symp. on the Conditioning of Radioactive Wastes for Storage and Disposal*, Utrecht, The Netherlands, (IAEA Vienna) pp. 375-390, 1982.

⁴A.G. Solomah, *Nuclear Technology*, Vol. 67, p. 311, 1983.

⁵V.M. Oversby and A.E. Ringwood, *Radioactive Waste Management Nuclear Fuel Cycle*, Vol. 2, p. 223, 1982.

by a constant. We can plot L_{inf} in the same form as in Figures 1 & 2 with the MCC-1 (or prorated value) as the intercept on the ordinate.

The instantaneous leach rates for glass are relatively constant; therefore, the 28-day MCC-1 values for glass, e.g., BSG or ASG (but probably not ASGC) would be the values to use for the constant phase. The BSG bulk rate previously recommended for the PA is $0.4 \text{ g/m}^2\text{-d}$ based on MCC-1. A typical forward rate of $1.5 \text{ g/m}^2\text{-d}$ was used by Grambow et al.² as shown in Table I with other data used to predict Fig. 1 results. The 28-day value should be better for the Step 1 constant value than using the (initial) forward rate based on an extrapolation to time zero. The latter represents an instantaneous maximum and not a constant rate over the first few days or months.

Table I (Table 5, p. 714, Ref. 1)

Parameters and data used in the performance analysis of borosilicate glass

(1) Materials properties:	
forward rate	$1.5 \text{ (g m}^{-2} \text{ d}^{-1})$
long-term rate	$0.0025 \text{ (g m}^{-2} \text{ d}^{-1})$
log K for silica saturation	-2.94
diffusion coefficient of SiO_2 in the surface layer	$10^{-8} \text{ (m}^2/\text{d)}$
selection of glass alteration phases	(JSS 1987)
temperature	$90 \text{ (}^\circ\text{C)}$
(2) Production related waste form data:	
geometric surface area	$1.71 \text{ (m}^2)$
factor for surface area increase	10
canister filling height	1.08 (m)
glass block diameter	0.42 (m)
glass density	$2.75 \times 10^3 \text{ (kg/m}^3)$
total mass	412.5 (kg)
molecular mass of the glass	61 (g/mol)
void volume in the container	$0.02 \text{ (m}^3)$
(3) Repository conditions:	
ground water flow rate	$0\text{--}10^{-5} \text{ (m}^3/\text{h)}$ $(0\text{--}100 \text{ (} \ell/\text{yr)})$
glass surface area to solution volume ratio (S/V)	$853 \text{ (m}^{-1})$

Measured long-term leach rates for glass are not available. Table I indicates the bulk leach rate value used by Grambow et al. in a performance analysis of BSG was $0.0025 \text{ g/m}^2\text{-d}$.

The Step 1 period would last through the knee of the curve in Fig. 2 for the Si curve, which is similar in time to that for the other high-flow rate curves. The time t_1 is then approximately one year. Step 2 then runs at times greater than one year.

A method to relate the cumulative mass fraction release rate to the bulk leach rates is given in the appendix. The results are presented here. For Stage 2, 99% of the fractional release of Si in Fig. 2 (assuming Si represents the bulk leach rate) is accomplished over 20,000 years. For BSG at 2500 kg/m^3 , the characteristic thickness is 7.37 m (a volume per unit surface area). This is reasonable if the waste repository is greater than or equal to 7.34 m thick. Applying the characteristic thickness to Stage 1 for 1% release over 1 year yields a leach rate of $0.505 \text{ g/m}^2\text{-d}$. The latter value is very close to the value of $0.4 \text{ g/m}^2\text{-d}$ originally recommended for the PA calculations; therefore,

the methods are consistent with the data.

Conclusions and Recommendations

The recommended dissolution rates for BSG glass are then:

- Step 1: 0.4 g/m²-d for t = 0 to 1 year.
Step 2: 0.0025 g/m²-d for t > 1 year.

Similar data is not available to determine the appropriate dissolution leach rates for ASG or for the ceramics. Based on the results with BSG, it is reasonable to assume that the Step 1 leach rate is the previously recommended MCC-1 value. The Step 2 values can then be calculated from the ratio of the ASG (or ceramics) Step 1 to BSG Step 1 value times the BSG Step 2 value, assuming the values are proportional.

The recommended ASG and ceramic values as calculated for the two-step model are given in Table II, along with the BSG values. The availability factor should be employed for the ceramics over both Steps 1 and 2.

Table II. Recommended two-step model leach-rate values in g/m²-d.

<u>Final Waste Form</u>		<u>Step1</u>	<u>Step2</u>
		0<t<1y	1<t<20,000+yr
Least Favorable Glass	BSG	0.4	0.0025
Most Favorable Glass	ASG	0.07	0.00044
Least Favorable Ceramic	IEB	0.08	0.00050
Most Favorable Ceramic	IEB4	0.05	0.00031

The major error in the assumptions for the ASG, IEB, or IEB4 final waste form is the lack of knowledge of the slope of the curves equivalent to those for BSG in Figures 1 & 2. The slope parameters are obtained through the assumption of proportionality to the BSG values. This will influence the time for the "100%" fractional release, which for Step 2 leach rates that are less than the BSG value of 0.0025 g/m²-d will exceed 20,000 years, by a factor of 10 or more.

Note that though the 100% release rate mass fraction was used to calculate the slope for the 1-10% region in the appendix, the time for the 100% release is a linear extrapolation. The actual time for a 100% release is hypothetical.

APPENDIX

Plotting formats of interest are:

$$\text{Linear: } y = a + bx \quad (1)$$

$$\text{Semi-log: } y = a + b \log x \quad (2)$$

$$\text{Log-Log: } \log y = a + b \log x \quad (3)$$

where a is the intercept and b is the slope of the line plotted.

The plot for Step 2 in Figs. 1 & 2 is Log-Log; hence,

$$\log m_{r1} = a + b \log t = \log a_2 + \log t^b = \log a_2 t^b \quad (4)$$

From which at times t and t_1 :

$$\frac{m_{r1}(t)}{m_{r1}(t_1)} = (t/t_1)^b \quad (5)$$

showing that the cumulative mass fraction is proportional to the ratio of times to the power of the slope.

The differential mass fraction leached can be related to the normalized leach rate as follows.

$$m_s dm_{r1} = L_{nrf} (m_i/m_s) dA dt \quad (6)$$

where the mass of the sample, m_s , is used to give the fractional mass release the dimensions of mass. Assuming that the surface area A is constant for leaching from a "slab" shaped repository, and that L_{nrf} is based on the initial m_i/m_s at t_0 , then integration from t_1 at the knee of the curve in Fig. 2 to t_2 at a known m_{r1} , in this case 100%, yields:

$$\begin{aligned} m_{r1}(t_2) - m_{r1}(t_1) &= (A/M_s) (m_i/m_s) \int L_{nrf} dt \\ &= (A/M_s) (m_i/m_s) L_{nrf,ave} (t_2 - t_1) \end{aligned} \quad (7)$$

Where $L_{nrf,ave}$ is the average leach rate over Step 2.

Note that the species leach rate (un-normalized) is just:

$$L_{nr} = (m_i/m_s) L_{nrf} \quad (8)$$

but for the bulk, $m_i = m_s$; hence,

$$L_{nr} = (1) L_{nrf} \quad (9)$$

Rearranging (7) to determine $L_{nr,ave}$ gives:

$$L_{mf,ave} = L_{nmf,ave} = \frac{m_s (m_{r1}[t_2] - m_{r1}[t_1]) (V)}{A (t_2 - t_1)} \quad (V)$$

$$L_{mf,ave} = \frac{\rho_s L_c (m_{r1}[t_2] - m_{r1}[t_1])}{(t_2 - t_1)} \quad (10)$$

where (V/V) is inserted to use the FF density, ρ_s , and the concept of characteristic length, $L_c = V/A$.

Solving for the BSG Step 2 with $t_1 = 1$ y, $t_2 = 20,000$ y, and the respective mass fraction releases as 1% and 100%, a BSG density of 2500 g/m^3 gives

$$L_{mf,ave} = 0.000339 \text{ g/m}^2\text{-d for } L_c = 1 \text{ m}$$

$$L_{mf,ave} = 0.00339 \text{ g/m}^2\text{-d for } L_c = 10 \text{ m}$$

and for the Grambow, et al.² value of $0.0025 \text{ g/m}^2\text{-d}$, the characteristic length is between 1 and 10 m at 7.37 m. This value corresponds to the "minimum" thickness of the layer of waste in the repository. It is in a reasonable range, giving credence to the calculations.

Extending (10) to Step 1, from $t_0 = 0$ with no mass release to t_1 with the other properties from above gives the mean leach rate over Step 1:

$$L_{mf,ave} = 0.505 \text{ g/m}^2\text{-d}$$

This value is very close to the previously and still recommended MCC-1 value for BSG for the PSPI PA calculations.

Prediction of borosilicate nuclear waste glass (JSS-A) corrosion as a function of exposure time. Effect of flow rate on the release of Si, B, Al and Fe (deionized water, 90°C). The fractional release is the amount of the respective element in solution relative to that in the glass (for waste form dimensions see table 1).

Fig. 1 (Fig 1, p. 715, Ref. 1)

Fig. 2 is Fig 1 modified for the 2-step process. Time t_0 is at time 0, t_1 is at the knee or junction of the two straight lines, and t_2 is at $t(100\%)$.

Prediction of borosilicate nuclear waste glass (JSS-A) corrosion as a function of exposure time for different release mechanisms. (a) Controlled by forward rate (table 1), (b) controlled by long-term rate (table 1), flow rate is 1 ℓ /yr, (c) as in (b) however the long-term rate increased by a factor of two and amorphous silica saturation allowed, (d) long-term rate equals zero, i.e., release is flow-rate controlled, (e) as in (d) but the near field effect considered (see section 2.1.1).

Fig. 3 (Fig 2, p. 716 Ref. 1)

Leach results for cesium and strontium from Synroc-C (10 wt% simulated HLW) and waste glass (PNL 76-68) at 100°C; leachant: deionized water, replaced daily (Reeve et al. 1982).

Fig. 4 (Fig. 18, p. 263, Ref. 1)

Differential leach rates of barium and cesium from Synroc-C prepared by cold-pressing and sintering (10 wt% simulated HLW) (Solomah 1983).

Fig. 5 (Fig. 15, p. 260, Ref. 1)

Effect of simulated HLW loading on cesium release from inactive Synroc-C [data plotted from Oversby and Ringwood (1982)].

Fig. 6 (Fig. 28, p. 273, Ref. 1)

Appendix H

Detailed Comparison of Assumptions and Parameter Values Used in Performance Assessments and Risk Assessments Related to the RWMC Subsurface Disposal Area

Appendix H

Detailed Comparison of Assumptions and Parameter Values Used in Performance Assessments and Risk Assessments Related to the RWMC Subsurface Disposal Area

This appendix presents detailed comparisons of the assumptions and parameter values used in six PAs or risk assessments, including this study, related to the Radioactive Waste Management Complex (RWMC) Subsurface Disposal Area (SDA) (see Table H-1 at end of this appendix). The table containing the comparisons was compiled originally by a panel of five personnel, which included at least one participant from each assessment cited. The entries underwent several rounds of comment and revision by the panel members. Additional changes were made by authors of this report after Phases 2 and 3 of the study.

The six SDA-related assessments^a whose assumptions and parameter values were compared are listed across the top of the table in columns 2 through 7. The five other assessments are Maheras et al. (1994), Loehr et al. (1994), Maheras et al. (1992), Rood (1992), and Rood (1993).

The left side (the first column) of the table lists the approximately 130 parameters, assumptions, and process models on which the studies were compared. The items are organized into the following categories:

- General (e.g., objective of assessment, type of waste addressed)
- Radionuclide inventory
- Waste characteristics (e.g., final waste form, release mechanisms, containers)
- Disposal unit characteristics (e.g., disposal depth, cover details)
- General timeline assumptions
- Undisturbed-waste scenario
 - General (e.g., pathways evaluated, receptors)
 - Groundwater pathway details
 - Air pathway details

a. The assumptions and parameters values used in this study for Sites B, C, D, and E are listed for informational purposes only—not for comparison against the other five assessments which address the SDA only.

- Biotransport pathway details
- Exposure route details
- Intrusion scenarios
- Regulatory standards used to compare calculated doses
- Results.

For a given parameter or assumption, one can compare the six assessments by scanning down the left side until the item is found, then reading across horizontally. The assumptions used by the various assessments are indicated where applicable in columns 2 through 7. Some of the assessments did not evaluate certain pathways or scenarios, so there are fewer than six entries in some rows.

Numerous reference sources are called out in the table. These references are not given in this appendix. The interested reader should consult the assessment that cited the reference.

Two more columns appear on the right side of Table H-1. Their purpose was to help ensure that, for all important parameters, this study either (a) used single-point values that were as conservative as those in the majority of the other assessments, or (b) used a range of values including conservative values.

One of the last two columns is "degree of conservatism in the ALLW PA." This column gives a simplified, qualitative indication of how conservative the evaluations were, relative to the other five assessments. The entries in this column were based on comparing the entries in each row and reflect the judgment of the panel members. For each entry, the alpha low-level waste (ALLW) performance assessment (PA) was concluded to be conservative or nonconservative or to have used typical values, compared with the values in the five other assessments.

It is stressed that the comparisons were not for the purpose of declaring one value "right" and another value "wrong." Such conclusions are generally not appropriate for several reasons. First, among informed peers, a range of values can be considered acceptable for most parameters used in PAs. Second, the studies were performed for different purposes, with different intended degrees of conservatism, in different regulatory environments, and for different portions of the waste. The question here focused more on how conservative the ALLW PA value was, compared with the range of acceptable values. Thus, the comparisons presented here simply provide a perspective on the values that could be used for a given parameter. They do not imply that the value of a parameter used in a given assessment is in error.

The last column — "spread, sensitivity, and importance"—provides a simplified ranking of the importance of each parameter. The term "spread" represents the range of values used for a given parameter in the six assessments. For each parameter, a measure of the spread (3 = large, 2 = moderate, 1 = small) was agreed upon by the panel members. The measure was based on considering the size of the range in values, compared with the full range that might have been chosen.

The term "sensitivity" is a measure of the extent to which the calculated doses depend on the parameter. Since the assessments considered different waste at different times, etc., there is no rigorous, single answer for the sensitivity. The results have different dependencies at different times for different radionuclides. Therefore, the simplified sensitivity ratings represent a collective judgment by the panel members based on (a) limited sensitivity studies in the six assessments, (b) experience with parameter sensitivities in other evaluations, and (c) professional judgment. Sensitivity was rated as 3 = high, 2 = moderate, 1 = low.

The "importance" of a parameter was assumed to be determined by the spread and the sensitivity. That is, if the assessments used widely differing values of a parameter, and if the sensitivity of the result to changes in the value of the parameter was high, then the parameter was judged to be important. In this simple scheme, the importance was calculated by multiplying the spread rating by the sensitivity rating. The maximum importance rating was $3 \times 3 = 9$. The minimum was $1 \times 1 = 1$.

References for Appendix H

- Loehr, C. A., B. H. Becker, D. E. Burns, R. M. Huntley, S. M. Rood, P. Sinton, and T. H. Smith, 1994, *Preliminary Scoping Risk Assessment for Waste Pits, Trenches, and Soil Vaults at the Subsurface Disposal Area, Idaho National Engineering Laboratory*, EGG-WM-11181, EG&G Idaho, Inc., May.
- Maheras, S. J., A. S. Rood, and R. Shuman, 1992, "A Performance Assessment of Radioactive Waste Encapsulated in Concrete and Disposed at the INEL," engineering design file, IWPF-0028, Appendix 2, EG&G Idaho, Inc., September.
- Maheras, S. J., A. S. Rood, S. O. Magnuson, M. E. Sussman, and R. N. Bhatt, 1994, *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment*, EGG-WM-8773, EG&G Idaho, Inc., May.
- Rood, A. S., 1992, "A Performance Assessment of Radioactive Waste Encapsulated in Iron-Enriched Basalt and Disposed at the Idaho National Engineering Laboratory's Radioactive Waste Management Complex," *Proceedings of the International Topical Meeting on Nuclear and Hazardous Waste Management, Spectrum '92*, engineering design file, IWPF-0028, Appendix 3, EG&G Idaho, Inc., August.
- Rood, A. S., 1993, "A Limited Performance Assessment for the Idaho Waste Processing Facility: 1993 Update Using Revised Parameters and Limited Sensitivity Analysis," engineering design file, IWPF-0044, EG&G Idaho, Inc., September.

Table H-1. Comparison of assumptions and parameters used in SDA-related assessments.

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
A. GENERAL								
A.1. Purpose	Early scoping of FWF performance at various sites	Scoping	Scoping	Scoping	Obtain approval from DOE-HQ	Preliminary scoping risk assessment to identify information gaps concerning potential hazards posed by wastes buried in the SDA; this information is intended to support planning of the Remedial Investigation under Comprehensive Environmental Response, Compensation and Liability Act (CERCLA)	NA	NA
A.2. Regulatory context	DOE Order 5820.2A and referenced regulations such as 40 CFR 141	No regulatory mandates	No regulatory mandates	No regulatory mandates	DOE Order 5820.2A and referenced regulations such as 40 CFR 141	No regulatory mandate, but the analysis was performed in accordance with EPA guidance documents; addressed radiological and nonradiological contaminants	NA	NA
A.3. Type of waste addressed	Mixed alpha LLW that is currently stored at RWMC TSA	Retrieved transuranic (TRU) waste and intermixed LLW that is currently buried at RWMC SDA	Retrieved TRU waste and intermixed LLW that is currently buried at RWMC SDA	Alpha LLW currently stored at RWMC TSA	LLW that is currently buried at RWMC SDA	Buried TRU waste and low-level waste, much of which is mixed, at RWMC SDA	NA	NA
A.4. Years generated	1970 to 1992	Unknown, believed to be the same inventory analyzed in the IEB PA (TRU inventory buried between 1954 and 1970)	TRU inventory buried between 1954 and 1970	Some of the waste is future generation and some is on the TSA pad or buried in a pit or trench	1984-2020 (projected inventory from 1994 on)	1952-1983	NA	NA
A.5. Years disposed of (conceptual model)	One instantaneous disposal in 1993	One instantaneous future disposal of treated waste; no decay of inventory	One instantaneous future disposal of treated waste; no decay of inventory	Future disposal; no decay of inventory.	1984-1993 projected: 1994-2020	1952-1983	NA	NA
A.6. Other					DOE Order 5820.2A requirement deals only with waste disposed of after 1988		NA	NA
B. RADIONUCLIDE INVENTORY								

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
B.1. Source of inventory	RWMIS data, as decayed to 1993 by Atwood and Schlafman (1993); additional refinements based on data for individual shipments in RWMIS and on limited waste information not in the shipping records	RWMIS and that derived from private sector participation (PSP)	RWMIS and that derived from PSP	RWMIS and that derived from Joe Gale	RWMIS for historical disposals, combination of generator projections and RWMIS extrapolations for future disposals	HDT used to generate P-CIDRA, on which PSRA is based	NA	Spread small, sensitivity high, importance 3/9
B.2. Initial Screening (explain)	Actinides+ progeny: (a) <0.5 Ci deleted except 0.1Ci U-238; and (b) <0.4 Ci of U-232, <0.1 Ci of U-234, and <0.1 Ci of Pu-242 deleted; others: <1 Ci deleted; and 13 Ci of Ce-144, 55 Ci of Cs-134, 3.4 Ci of Kr-85, 1.79 Ci of Ni-63, 14 Ci of Pm-147, and 14 Ci of Pr-144 deleted; validity of initial screening confirmed in later detailed screening and detailed intruder calculations; in nonactinide inventory with additional refinements C-14, I-129, and Tc-99 were included even though their activities were <1 Ci each	Unknown	Unknown	Unknown, fission products deleted.	None, all inventory considered initially. Actinides where parent + progeny have $t_{1/2}$ <10 yr deleted; short-lived progeny + long-lived parent resulting in <1 nCi deleted, all short-lived daughters of long-lived parents included	(a) Used disposed upper bound inventory estimates for screening; (b) kept radionuclides with half-life >1 yr and >1 Ci; (c) kept radionuclides whose daughter has half-life >1 yr and >1 Ci; (d) kept daughter radionuclides with half-lives >1 yr, >1 Ci, and for which the slope factor of the parent does not account for the daughter effects; (e) for remaining radionuclides, annual decay to 1994 was performed, and retained radionuclides with activity >1 Ci; (f) conservative risk calculations were used to further screen out contaminants with risk $<10^{-7}$ and hazard quotient <0.1	Assumptions confirmed by later detailed screening	Spread small, sensitivity varies, importance 1/9 to 3/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
B.3. Specification of wastes considered	Remote-handled (RH) and contact-handled (CH) ALLW considered. Assumptions for most runs: (a) activity is uniformly distributed, (b) pretreatment activity = posttreatment activity, (c) baseline case has alpha activity = 100 nCi/g; some runs performed with a "refined" inventory that considered CH and RH fractions separately; additional runs performed with higher U-233 and Th-232 inventories	Unknown. See A.3	Unknown. See A.3	CH and RH waste considered	CH and RH waste considered	All waste considered	For most runs, conservative for CH waste, non-conservative for RH waste; for runs with refined inventory, typical modeling assumptions	Spread moderate, sensitivity moderate, importance 4/9
B.4. Handling of decay from time of generation to start time of PA	Decay + ingrowth from generation to start of PA considered by using decayed values from Atwood and Schlafman (1993) or from calculated decay	No decay accounted for in the reported inventory	No decay accounted for in the reported inventory	Decay is accounted for before treatment and burial	Decay was considered for each year of disposal (36 yr)	Annual decay with ingrowth	Typical modeling assumption	Spread small, sensitivity high, importance 3/9
B.5. Handling of MAP and MFP	MFP is 50% Sr-90 and 50% Cs-137; <1 Ci of MAP screened out	Not discussed	Not discussed	NA	MAP and MFP as 50% Sr-90 and 50% Cs-137	Radioactivity previously listed under generic names was distributed in the HDT on a waste-stream-specific basis	Realistic simplified modeling assumption	Spread small, sensitivity low, importance 1/9
B.6. Handling of unidentified beta-gamma	Unidentified beta-gamma activity is 50% Sr-90 and 50% Cs-137	Not discussed	Not discussed	NA	Unidentified beta-gamma activity is 50% Sr-90 and 50% Cs-137	Radioactivity previously listed under generic names was distributed in the HDT on a waste-stream-specific basis	Typical modeling assumption	Spread small, sensitivity low, importance 1/9
B.7. Handling of unidentified alpha	None in inventory	None in inventory	None in inventory	Unknown	None in inventory	Radioactivity previously listed under generic names was distributed in the HDT on a waste-stream-specific basis	NA	NA
B.8. Projection of future wastes	Only stored waste considered	None in inventory	None in inventory	None in inventory	Generator forecasts and extrapolated averages of historical disposals	None in inventory	NA	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Lochr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
B.9. Other	The objective of developing the simplified inventory was to project and retain the top contributors to dose, not to produce a comprehensive inventory				Assigning MAP, MFP, unidentified beta-gamma based on activity distribution in wastes received from 1984-1993 was performed in uncertainty analysis with negligible effect on estimated doses		NA	NA
C. WASTE CHARACTERISTICS								
C.1. Disposed of with or without treatment	With treatment; limited runs with untreated waste also performed	With treatment	With treatment	With or without treatment	Without treatment	Without treatment.	NA	NA
C.2. Type of waste treatment	Thermal treatment to meet LDR. Cement (PCC, FUETAP), glass (BSG, ASG), and ceramic (IEB, IEB4) FWFs evaluated	55-gal drums solidified with concrete	Vitrification in basalt	None	None	None	NA	NA
C.3. Final Waste Forms	Cement (PCC, FUETAP), glass (BSG, ASG), and ceramic (IEB, IEB4) FWFs evaluated. Limited runs with untreated waste	Concrete	IEB	Untreated waste	Untreated waste	Untreated waste	NA	NA
C.4. Degradation of Treated Waste Form	Cement fracturing simulated by shorter diffusion path. Glass and ceramic fracturing simulated by increasing FWF surface area	Cement fracturing simulated by increase in porosity	Not considered	NA	NA	NA	ALLW most conservative among studies involving treated waste	Spread large, sensitivity medium, importance 6/9
C.5. Release mechanisms	Cement release by diffusion, glass, and ceramic release by dissolution; unrelated waste release by surface washoff and, alternatively, by diffusion	Diffusion from nondegraded concrete and advection from degraded concrete	Dissolution of the blocks	Sorption leaching	Sorption-limited leaching, corrosion of activated metals and diffusion	Corrosion release of activation products from metals; surface washoff for other contaminants	Appropriate approach	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
C.6. Credit for waste containers	None	None	None	None	Some credit was taken for waste contained in metal containers (10-yr mean lifetime); no credit was taken for cardboard or wooden boxes	Assumed instantaneous failure at beginning of release (1994) for all containers/forms except steel drums, which were decayed according to the INEL-specific drum decay curve	More conservative than RWMC PA and PSRA.	Sensitivity varies depending on nuclide inventory; importance 2/9 to 6/9
D. DISPOSAL CHARACTERISTICS DIRECTLY RELATED TO DISPOSAL								
D.1. Location of disposal site	5 sites: RWMC SDA Site 14 (INEL), Area G (NTS), E-Area Vaults Disposal Facility (SRS), and 200 West Burial Grounds (Hanford)	RWMC SDA	RWMC SDA	RWMC SDA	RWMC SDA	RWMC SDA	NA	NA
D.2. Type of disposal unit(s)	Pits. Vaults for SRS only	Pits	Pits	Pits	Pits and soil vaults	Pits, trenches, and soil vaults	NA	NA
D.3. Dimension of disposal unit, parallel and perpendicular to groundwater flow direction	51.8 m × 51.8 m (FUETAP/PCC) 31.5 m × 31.5 m (IEB4) 34.1 m × 34.1 m (IEB) 51.3 m × 51.3 m (BSG) 36.2 m × 36.2 m (ASG)	160 m × 160 m	77 m × 77 m	76 m × 76 m	257 m × 122 m	Rectangle of 181 × 647 m oriented with the long edge perpendicular to groundwater flow in the Snake River Plain Aquifer	RWMC PA and PSRA modeled actual layout. Others were hypothetical	Spread moderate, sensitivity moderate, importance 4/9
D.4. Depth of top of waste stack (belowgrade)	1.5 m and 2.7 m	4.6 m (this value was chosen to prevent the intruder from excavating the waste during basement construction)	3-m (this value was chosen to prevent the intruder from excavating the waste during basement construction)	0 m	Pits: 2.4 m soil vaults: 3.3 m	1.8 m (average thickness of soil cover)	Some more, some less; typical assumptions used	Spread big, sensitivity high, importance 9/9
D.5. Thickness of waste layer	6.9 m	1.6 m	2.4 m	2.7 m	Pits: 6.1 m, soil vaults: 3.0 m	2.53 m (average thickness of waste based on thickness in units)	NA	NA
D.6. Mixing of waste with backfill	Waste and backfill homogeneously mixed	Waste was not mixed with backfill; a 1-m thick soil layer underlying the waste blocks was accounted for	Waste was not mixed with backfill; a 1-m thick soil layer underlying the waste blocks was accounted for	Waste homogeneously mixed with backfill	Most wastes mixed homogeneously with backfill, activated metals and beryllium reflector blocks remain separate but release into the total backfilled soil	Model assumed homogeneously mixed with soil; actual conditions are variable	Typical modeling assumption	Spread medium, sensitivity low, importance 2/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
D.7. Cover installed (y/n)	No for undisturbed performance; yes for intrusion scenarios	No	No	No	Yes. Cover considered for reduction of infiltration during institutional control period (2020-2120)	Yes	Typical modeling assumption	Spread big, sensitivity low to high, importance 3/9 to 9/9
D.8. (if y) Degradation of cover	Instant, total erosion at year 500	NA	NA	NA	Pits: 2-m operational cover has 3-m closure cover added; erodes to a total thickness of 2.4 m soil vaults: 5 m cover erodes to 3.3 m thickness	No degradation assumed	Typical modeling assumption	Spread medium, sensitivity medium, importance 4/9
D.9. Other					Dimensions of disposal unit approximately match the combined area of Pits 17, 18, 19, and 20		NA	NA
E. GENERAL TIMELINE ASSUMPTIONS								
E.1. Date waste emplacement operations end	1993	1970	1970	Future disposal, unknown	Conservative date of 2020 was used	1983	NA	NA
E.2. Period of institutional control	100 yr	100 yr	100 yr	NA (no intruder scenarios considered)	100 yr	100 yr	Typical assumption following regulatory guidance	NA
E.3. Start of modeling vs. start of waste emplacement	Instantaneous waste emplacement at start of simulation	Instantaneous waste emplacement at start of simulation	Instantaneous waste emplacement at start of simulation	Instantaneous waste emplacement at start of simulation	The start of modeling coincided with the start of waste emplacement: 1984 to 2020	Modeling starts at beginning of 1994; start of waste emplacement in 1952	Typical modeling assumption	Spread low, sensitivity low, importance 1/9
E.4. Other					Soil vault inventories were considered to be mixed in the pit volume for groundwater pathway calculational purposes		NA	NA
F. UNDISTURBED WASTE SCENARIOS								

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.1. GENERAL								
F.1(a) Transport Pathways	Groundwater, airborne (radon assumed to be only important airborne pathway, based on RWMC PA)	Groundwater	Groundwater	Groundwater	Groundwater, airborne, radon	Groundwater, air, soil, direct radiation	Typical modeling assumption	NA
F.1(b) Receptors	Located 100 m and 215 km from the downgradient edge of the disposal unit; onsite for intrusion scenarios	Receptor located at edge of source for evaluation of groundwater pathway	Receptor located at downgradient edge (38.5 m from center) of source for evaluation of groundwater pathway	Receptor located at downgradient edge of source for evaluation of groundwater pathway	100 m downgradient from Pits 17-20 boundary and at the site boundary (5,500 m downgradient during institutional control, 100 m downgradient thereafter)	Onsite future worker and onsite future resident located at downgradient edge of combined source	Typical or slightly non-conservative (100 m)	Spread medium, sensitivity medium, importance 4/9
F.1(c) Exposure Pathways	Ingestion, inhalation	Ingestion	Ingestion	Ingestion, inhalation, transfer to food via irrigation (all pathways)	Consumption and irrigation with groundwater	Onsite future worker: fugitive dust inhalation, soil ingestion, dermal contact with soil, and external radiation exposure at surface; onsite future resident: fugitive dust inhalation, ingestion of soil, ingestion of groundwater, meat, dairy products, and crops, dermal contact with soil, and external radiation exposure at surface	Less conservative than RWMC PA and PSRA, same as others	Spread medium to high, sensitivity low to medium, importance 2/9 to 6/9
F.1(d) Exposure Scenarios	Groundwater = leaching and transport to receptor well, drinking water and irrigation of crops; air = inhalation of radon	Drinking water	Drinking water	All pathway (drinking water, ingestion of contaminated foods)	All pathway (drinking water, ingestion of contaminated foods), groundwater, inhalation of airborne dust, external exposure	Groundwater → ingestion (resident only); biotic transport → suspension → air → inhalation; biotic transport → direct contact → soil → ingestion/contact; biotic transport → radiation → external exposure; radiation through cover → external exposure		

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2. GROUNDWATER PATHWAY DETAILS								
F.2(a). GENERAL								
F.2(a).1. Transport pathways	Waste zone → vadose zone → saturated zone	Waste zone → vadose zone → saturated zone	Waste zone → vadose zone → saturated zone	Waste zone → vadose zone → saturated zone	Waste zone → vadose zone → saturated zone	Waste zone → vadose zone → saturated zone	Same assumptions	NA
F.2(a) 2. Period of performance evaluated	Time to maximum dose	Time to maximum dose	Time to maximum dose	Time to maximum dose	10,000 yr from 1984, doses from nuclides, with a peak dose after that time, were presented in tables for informational purposes only	1,000 yr with 30-yr averages for 10 different periods; also calculated peak risk resulting from peak concentrations in groundwater regardless of time	Typical and comprehensive	Spread small, sensitivity medium, importance 2/9
F.2(a).3. Groundwater transport screening (y/n, if y explain)	Yes, used Seitz method, function of inventory, K_d , half life, dose conversion factor; parents of chains not screened	None	None	None	Yes, GWSCREEN was used to calculate a screening dose using more conservative transport parameters and assumptions	Yes, see B.2.	Typical modeling assumptions used	Spread large, sensitivity varies, importance 3/9 to 6/9
F.2(a).4. Decay and ingrowth	Accounted for at the receptor well; progeny travel with parent	Accounted for at the receptor well; progeny travel with parent	Accounted for at the receptor well; progeny travel with parent	Accounted for at the receptor well; progeny travel with parent	Accounted for at the receptor well; progeny travel with parent	Ingrowth during transport away from waste zone, progeny travel with parent	Conservative approach	Spread small, sensitivity high, importance 3/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(b). HYDROLOGIC PROCESSES OR PARAMETERS								
F.2(b).1 Infiltration Rate	RWMC SDA = 1.32, 5, and 10 cm/yr; Site 14 = 1.2, 5, and 10 cm/yr; NTS = 0.6 cm/yr; SRS = 40, 80, and 120 cm/yr; Hanford = 0.5, 5, and 10 cm/yr		5 cm/yr	4 and 10 cm/yr	4 to 10 cm/yr, 7 cm/yr were used for compliance determination purposes	10 cm/yr	Non- conservative for most runs, typical for runs with high infiltration rates	Spread high, sensitivity high, importance 9/9
F.2(b).2 How derived	Infiltration assumed to be 6% of precipitation (except values of 5 and 10 cm/yr at RWMC and Site 14); SRS value of 40 cm/yr from site data, plus 80 and 120 cm/yr for sensitivity; Hanford value of 0.5 cm/yr from site data plus 5 and 10 cm/yr for sensitivity	From Baca et al. 1992, assumed value equal to 20% of precipitation	From Baca et al. 1992, assumed value equal to 20% of precipitation	Same as 1994 RWMC LLW Radiological PA	In situ moisture content and hydraulic characterization of samples from 240-ft interbed beneath the RWMC coupled with unit gradient assumption (Magnuson and McElroy 1993); further relied on infiltration estimates from neutron access tube monitoring inside the SDA (McElroy 1993)	Upper end of range given in Magnuson and McElroy, 1993	NA	NA
F.2(b).3 Cover/Engineered Barrier (y/n)	No, except for SRS facility, which has concrete vault and cover	No	No	No	Yes, maintained thick soil cover during institutional control had a reduced infiltration rate of 1 cm/yr based on simulation study by Magnuson (1993)	Yes	Conservative compared to RWMC PA, consistent with others	Spread medium, sensitivity high, importance 6/9
F.2(b).4 (if y) Assumptions used in degradation of cover	Not applicable at INEL and NTS sites; concrete cover at SRS assumed to provide complete containment for 575 years	NA	NA	NA	Barrier was conservatively assumed to fail hydraulically during the 5 yr after institutional control ended; rapid failure was assumed due to high likelihood of rodent burrowing	No credit taken for cover with respect to the groundwater pathway	NA	NA
F.2(b).5 Thickness of sediment underlayer in disposal unit (credit taken?)	No underlayer	1 m of backfill sediments	1 m of backfill sediments	No underlayer	No underlayer	No underlayer	Typical and conservative	Spread medium, sensitivity medium, importance 4/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(b).6 Movement of infiltrating water through waste zone (y/n)	Yes	Yes	Yes	Yes	Yes	Yes	Typical	NA
F.2(b).7 Steady-state or transient	Steady-state	Steady-state	Steady-state	Steady-state	Steady-state	Steady-state	Typical and consistently non-conservative	NA
F.2(b).8 Constitutive relationship of flow in waste zone	Same as RWMC PA for the backfill, FWFs considered impermeable	From Baca et al. 1992 as was originally implemented in GWSCREEN, this implementation incorrectly represented residual moisture content, and, as a result, the estimated moisture contents differ from those for the RWMC LLW PA and the IWPF	From Baca et al. 1992 as was originally implemented in GWSCREEN, this implementation incorrectly represented residual moisture content, and, as a result, the estimated moisture contents differ from those for the RWMC LLW PA and the IWPF	Same as 1994 RWMC LLW Radiological PA	Geometric averaging of fitted van Genuchten curves from four surficial sediment samples hydrologically characterized and reported in McElroy and Hubbell (1990); the geometric averages were first reported in Baca et al. 1992 and were subsequently used as surficial soil default parameters in the GWSCREEN model (Rood 1994)	Same as RWMC PA	Typical and consistent approach	NA
F.2(b).9 Moisture content (if steady-state)	RWMC = 0.32 (1.32 cm/yr), 0.36 (5 cm/yr), 0.38 (10 cm/yr); Site 14 = 0.32 (1.2 cm/yr), 0.36 (5 cm/yr), 0.38 (10 cm/yr); NTS = 0.30; SRS = 0.178 (40 cm/yr), 0.19 (80 cm/yr), 0.21 (120 cm/yr)	0.285	0.285	0.32 & 0.34, for 0.04 to 0.10 m/yr infiltration	0.33 cm ³ /cm ³	0.3	Typical and consistent approach	Spread small, sensitivity medium, importance 2/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(b).10 Movement of infiltrating water in basalts within the vadose zone	Sediment and basalt simulated by varying K_d and porosity and bulk density; did not model fracture flow; some runs took no credit for basalt	Water travel time was accounted for only in the basalts	Water travel time was accounted for only in the basalts	Water travel time was accounted for only in the sediments	Water travel time was accounted for only in the sediments	Water travel time was accounted for only in the sediments	For most runs, ALLW PA was non-conservative; long vadose zone travel time due to assumed infiltration rate and moisture content; for runs taking no credit for basalt, typical	Spread high, sensitivity high, importance 9/9
F.2(b). 11 (if y) Thickness of basalt layers	RWMC = 167 m basalt under 9 m sediment; no basalt at Site 14, NTS, or SRS; Hanford basalt below water table	176 m	176 m	NA	NA	NA	NA	NA
F.2(b).12 Constitutive relationship for flow in basalt	Basalt characteristic curve from GWSCREEN	From Bishop (1991) basalt matrix characterization data as implemented in Baca et al. 1992; this same curve was assigned as the default basalt moisture characteristic curve in GWSCREEN; relies on simplistic assumptions to incorporate fracture flow effects	From Bishop (1991) basalt matrix characterization data as implemented in Baca et al. 1992; this same curve was assigned as the default basalt moisture characteristic curve in GWSCREEN; relies on simplistic assumptions to incorporate fracture flow effects	Not considered	Not considered	Not considered	NA	NA
F.2(b).13 Moisture content of basalt (if steady-state)	RWMC = 0.07; Site 14, NTS, SRS, and Hanford = NA	0.05	0.05	NA	NA	NA	NA	NA
F.2(b).14 Movement of infiltrating water in sedimentary interbeds (y/n)	All sediment and interbed layers lumped together	Not considered	Not considered	Yes	Yes	Yes	Typical	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(b).15 (if y) thickness of sedimentary interbeds	RWMC = 9 m; Site 14 = 80 m (sediment); NTS = 245 m (sediment); SRS = 21 m (sand and soil); Hanford = 75 m (sediment)	NA	NA	10.66 m, based on interim calculations for the RWMC LLW PA; deeper interbeds not considered	12.5 m, no credit taken for A-B interbed due to location of Pits 17-20; average Anderson and Lewis (1989) thicknesses for B-C, C-D, D-E, and E-F interbeds were used	16.46 m equal to the sum of the average interbed thicknesses given in Anderson and Lewis (1989), interbed thicknesses for the A-B, B-C, C-D, D-E, E-F, and F-G interbeds	Conservative	Spread small, sensitivity moderate, importance 2/9
F.2(b).16 Constitutive relationship for flow in sedimentary interbeds	Based on basalt characteristic curves from GWSCREEN	NA	NA	Same as 1994 RWMC LLW Radiological PA	Hydrologic characterization of core samples from 110-ft and 240-ft interbeds reported in McElroy and Hubbell (1990); these samples were grouped and averaged and representative van Genuchten moisture characteristic curves for both interbeds were determined (Magnuson and McElroy 1993)	Based on surficial sediment characteristic curves from GWSCREEN	NA	NA
F.2(b).17 Moisture content (if steady-state)	RWMC = 0.07; Site 14 = 0.06; NTS = 0.05; SRS = 0.074 (sand), 0.088 (soil); Hanford = (0.039)	NA	NA	0.27, as in RWMC LLW PA travel, this was the moisture content necessary to match cumulative estimated travel times across sedimentary interbeds	Varied by interbed: B-C 0.082; C-D 0.38; D-E & E-F 0.40; moisture content that was necessary to match cumulative vadose zone travel time was 0.168	0.3	ALLW PA is non-conservative compared with other analyses	Spread large, sensitivity medium to large, importance 6/9 to 9/9 (nuclide-specific)
F.2(b).18 Cumulative water travel time through vadose zone	Total (including waste zone) is RWMC = 933 yr for most runs; some runs made with travel times as short as 6.3 yr; Site 14 = 400 yr; NTS = 2,246 yr, SRS = 4.52 yr, Hanford = 560 yr	176 yr	176 yr	28.7-71.9 yr	For the 7 cm/yr infiltration rate, the cumulative water travel time from beneath the pit to the aquifer was 30 yr	49.4 yr	ALLW PA is nonconservative for most runs; is conservative for a limited number of runs	

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(b).19 Other Unsaturated Zone					(a) The infiltration rate at depth was not reduced even with a cover during institutional control based on an assumption of easy horizontal movement within the fractured basalts; (b) greater than unit gradients based on field measurements were used to estimate velocity across the C-D interbed; (c) travel times were estimated across individual interbeds and summed to determine cumulative water travel time		NA	NA
F.2(b).20 Aquifer average linear velocity	RWMC = 570 m/yr; Site 14 = 668 m/yr; NTS = 1.0, 0.1, and 0.01 m/yr; SRS = 6.67 m/yr; Hanford = 22.0 m/yr	570 m/y	570 m/y	890 m/yr This value was an interim RWMC PA value that was later changed to 560 m/y	560 m/yr, Wood (1989)	570 m/yr	Typical values used	Spread small, sensitivity low to medium depending on receptor well location; importance 1/9 to 3/9
F.2(b).21 Aquifer porosity	RWMC = 0.10; Site 14 = 0.10; NTS = NA; SRS = 0.4; Hanford = 0.1	0.1	0.1	0.1	0.10 cm ³ /cm ³ , Robertson (1974)	0.1	Typical values used	Spread small, sensitivity high, importance 3/9
F.2(b).22 Flooding events evaluated (y/h)	No	No	No	No	Yes, flooding was evaluated as a sensitivity case; floods of magnitude 20, 35, and 70 cm, occurring shortly after termination of institutional control, were evaluated	No	Only RWMC PA considered flooding, and that was as a sensitivity case; ALLW PA was non-conservative	Spread small, sensitivity medium to high, importance 3/9 to 6/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(b).23 Codes or methodology used	VS2DT for flow through disposal facility; water travel time through vadose zone estimated analytically as a function of the infiltration rate, moisture content, and vadose zone thickness; saturated zone had associated aquifer average linear velocity	Codes were written or modified (GWMOD) to perform calculations for this problem	Codes were written or modified (GWMOD) to perform calculations for this problem	GWSCREEN; water travel time through vadose zone estimated analytically as a function of the infiltration rate, moisture content, and vadose zone thickness	Source term: analytic unit-gradient solutions as described above; water travel time through vadose zone estimated analytically as a function of the infiltration rate, moisture content, and vadose zone thickness; saturated zone: GWSCREEN for both steady-state and flooding	DUST	Typical approach	NA
F.2(c). SOURCE ZONE TRANSPORT PROCESSES								
F.2(c).1 Include sorption (if y, cite sources)	No, except for some cases; these cases used 1/10 of the K_d of the vadose zone soils at the site; additional cases, representing tailored backfill, used the full value of the K_d of the vadose zone soils	Yes, Case et al. (1990)	Yes, Case et al. (1990)	Yes, RWMC PA (Case et al. 1990) and TRACK 2 guidance document (DOE 1993)	Yes, sorption coefficient data in the 1990 PA (Case et al.) were used; comparisons and sensitivity analysis were also performed using data from Sheppard and Thibault (1990), the TRACK 2 manual (DOE 1993), Baes and Sharp (1984), Del Debbio and Thomas (1989), and the NEA Database (Ticknor and Ruegger 1989)	Yes: DOE (1993), Sheppard and Thibault (1990), Baes and Sharp (1983), Coughtrey et al. (1987), Looney et al. (1987), Del Debbio and Thomas (1989), Hawkins and Short (1965), Schmalz (1972)	ALLW generally conservative	Spread high, sensitivity high, importance 9/9
F.2(c).2 Include dispersion	No	No	No	No	No, mixing cell model for source term	Yes	Consistent conservative approach	NA
F.2(c).3 Include diffusion	No	No	No	No	No	Yes		
F.2(c).4 Solubility limits	Yes - EQ3/6 limits used but never reached	No	No	Yes	Yes, but never reached	Yes	NA	NA
F.2(c).5 Codes or methodology used	DUST code	First-order kinetic model written for calculation of release from source term; BLT used for sorption controlled release from soil layer	First-order kinetic model written for calculation of release from source term	GWSCREEN	A custom code was written to solve for the time-dependent inventory and release rate from the source model	DUST	Codes appropriate for types of FWFs	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(d). UNSATURATED ZONE TRANSPORT PROCESSES								
F.2(d).1 Include sorption	Yes	Yes	Yes	Yes	Yes	Yes	Typical consistent approach	NA
F.2(d).2 Include dispersion	No	No	No	No	No, conservatively neglected, plug flow model assumed	No	Typical consistent approach	NA
F.2(d).3 Include diffusion	No	No	No	No	No	No	Typical consistent approach	NA
F.2(d).4 Sorption coefficients source	Case et al. (1990), DOE (1982), and DOE (1984); SRS values from EAVDF PA (1994), Hanford values from 200W Area PA (1994)	Case et al. (1990), DOE (1982), and DOE (1984)	Case et al. (1990), DOE (1982), and DOE (1984)	Case et al. (1990), DOE (1982), and DOE (1984)	Yes, sorption coefficient data in the DOE 1982 and 1984 studies were used; comparisons and sensitivity analyses were also performed using data from Sheppard and Thibault (1990), the TRACK 2 manual (DOE 1992), Baes et al. (1984) for Tc and K, Del Debbio and Thomas (1989), and the NEA Database	Yes, DOE (1993), Sheppard and Thibault (1990), Baes and Sharp (1983), Coughtrey et al. (1987), Looney et al. (1987), Del Debbio and Thomas (1989), Hawkins and Short (1965), Schmalz (1972)	Identical values used for all key nuclides (Pu, U, and Ra)	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(d).5 Codes or methodology used	1-D analytical solution; nuclide velocity calculated as product of the vadose zone water travel time and the nuclide-specific retardation coefficient for the site-specific geologic stratigraphy	1-D analytical solution; nuclide velocity calculated as product of the vadose zone water travel time and the nuclide-specific retardation coefficient for the site-specific geologic stratigraphy	1-D analytical solution; nuclide velocity calculated as product of the vadose zone water travel time and the nuclide-specific retardation coefficient for the site-specific geologic stratigraphy	1-D analytical solution; nuclide velocity calculated as product of the vadose zone water travel time and the nuclide-specific retardation coefficient for the site-specific geologic stratigraphy; implemented using GWSCREEN	1-D analytical solution; nuclide velocity calculated as product of the vadose zone water travel time and the nuclide-specific retardation coefficient for the site-specific geologic stratigraphy; implemented using GWSCREEN	GWSCREEN	Typical consistent approach	NA
F.2(e). SATURATED ZONE TRANSPORT PROCESSES								
F.2(e).1 Include sorption	Yes	Yes	Yes	Yes	Yes	Yes	Typical consistent approach	NA
F.2(e).2 Include dispersion	Yes	Yes	Yes	Yes	Yes	Yes	Typical consistent approach	NA
F.2(e).3 Include diffusion	Yes (NTS only)	No	No	No	No	Yes	Typical consistent approach	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.2(e).4 Source of information for sorption coefficients	Case et al. (1990), DOE (1982), and DOE (1984); SRS data from EAVDF PA (1994), Hanford data from 200W Area PA (1994)	Case et al. (1990), DOE (1982), and DOE (1984)	Case et al. (1990), DOE (1982), and DOE (1984)	Case et al. (1990), DOE (1982), and DOE (1984)	Sorption coefficient data in the DOE 1982 and 1984 studies were used; comparisons and sensitivity analyses were also performed using data from Sheppard and Thibault (1990), the TRACK 2 manual (DOE 1992), Baes et al. (1984) for Tc and K, Del Debbio and Thomas (1989), and the NEA Database (Ticknor and Ruegger 1989)	Yes: DOE (1993), Sheppard and Thibault (1990), Baes and Sharp (1983), Coughtrey et al. (1987), Looney et al. (1987), Del Debbio and Thomas (1989), Hawkins and Short (1965), Schmalz (1972)	Typical consistent approach	NA
F.2(e).5 Well screen thickness	15 m per Track 2 guidance	12 m	12 m	12 m	100 m receptor: 12 m (Seitz 1991); site boundary receptor: 76 m (Robertson 1974)	15 m	ALLW non-conservative (25% higher) than most others	Spread small, sensitivity medium, importance 2/9
F.2(e).6 Codes or methodology used	Analytical Green's function solutions to the advection-dispersion equation as presented in Codell et al. (1982) and implemented by IT	Analytical Green's function solutions to the advection-dispersion equation as presented in Codell et al. (1982) and implemented in modified GWMOD	Analytical Green's function solutions to the advection-dispersion equation as presented in Codell et al. (1982) and implemented in modified GWMOD	Analytical Green's function solutions to the advection-dispersion equation as presented in Codell et al. (1982) and implemented in GWSCREEN	Analytical Green's function solutions to the advection-dispersion equation as presented in Codell et al. (1982) and implemented in GWSCREEN	GWSCREEN	Typical approach	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Lochr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.3. AIR PATHWAY DETAILS								
F.3(a) Air pathways addressed	Rn-222 emission from the disposal facility; other air pathways were not pursued based on results from RWMC PA	None	None	None	Gaseous release of H-3 and C-14, and resuspension of contaminated surface soil	Resuspension	Consistent approach except for RWMC PA and PSRA which considered limited air pathway	Varies
F.3(b) Code(s) used	Flux of Rn-222 calculated through analytical equation cited in Case et al. (1990); concentration of Rn at receptor at the downwind edge of the disposal facility using the Eulerian box model (EPA 1991)	NA	NA	NA	GENII; source term for air pathway derived from biotic transport of waste to the surface	Methodology from DOE (1993)	NA	NA
F.4. BIOTRANSPORT PATHWAY DETAILS								
F.4(a) Pathways addressed	No	No	No	No	Yes	Burrowing animals and insects, plant uptake, and animal uptake	Consistent approach except for RWMC and PSRA, which considered biotransport	Varies
F.4(b) Code(s) used	NA	NA	NA	NA	Hand calculation	DOSTOMAN	NA	NA
F.4(c) Other	No animal intrusion was considered because the depth of the waste precludes animal intrusion				Transport by this pathway was considered only for some plant species; no animal intrusion was considered because the depth to the waste precludes intrusion by animals		NA	NA
F.5. EXPOSURE ROUTE DETAILS								

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
F.5(a) Exposure pathways	Ingestion (groundwater, crops, and animal products)	Groundwater ingestion	Groundwater ingestion	All pathways, drinking water	Inhalation, ingestion (direct ingestion of groundwater and all pathway), external exposure	Onsite future worker: fugitive dust inhalation, soil ingestion, dermal contact with soil, external radiation exposure; onsite future resident: fugitive dust inhalation, soil ingestion, groundwater ingestion, ingestion of meat, dairy products, and crops, dermal contact with soil, and external radiation exposure	ALLW PA considered more pathways than others except for RWMC PA and PSRA	Spread big, sensitivity low, importance 3/9
F.5(b) Shielding factor for external exposure	NA	NA	NA	NA	Two factors evaluated: 0.7 and 0.36 based on NRC (1977) and Konz et al. (1989), respectively	Used MicroShield v 4.0; no shield other than soil cover; factor not reported	NA	NA
F.5(c) Air immersion assumptions	NA	NA	NA	NA	Exposure to contaminated air for 1 yr using above shielding factors	NA	NA	NA
F.5(d) Diet assumed	Kennedy and Streng (1992) and Aaberg and Kennedy (1990); GENII default values of 2L/d, 365 d/yr; 25% vegetables home-grown	2 L/d, 365 d/yr for direct groundwater ingestion	2 L/d, 365 d/yr for direct groundwater ingestion	0.7 L/d water ingestion rate, 18 kg/yr leafy veg, 112 L/yr milk, 85 kg/yr beef, and 176 kg/yr produce	For the all-pathway analysis, the diet was taken from Yang and Nelson, NCRP, Rupp, Ng, EPA, and some site-specific data; the water ingestion rate was 0.7 L/d, beef ingestion was 85 kg/yr, milk ingestion was 0.3 L/d, produce ingestion was 176 kg/yr and leafy vegetables consumption was 18 kg/yr; 70% vegetables home-grown	Soil ingestion: resident adult 100 mg/d, resident child 200 mg/d, worker 50 mg/yr; water ingestion: resident 2 L/d, worker none; beef/dairy ingestion: resident 75 g/d beef and 300 g/d dairy, worker none; crops ingestion: resident 122 g/d, worker none	More conservative than IWPF and RWMC PAs	Spread medium, sensitivity medium, importance 4/9
F.5(e) Code(s) used	DITTY code of the GENII Package	Spread sheet	Spread sheet	Spread sheet to calculate an equivalent water ingestion rate; dose calculation performed in GWSCREEN	Spread sheet calculation for all pathway analysis and GENII for atmospheric dose calculations	DOSTOMAN, GWPOST	Appropriate approach	NA
F.5(f) Dose conversion factors	DOE/EH-0071	DOE 1988	DOE 1988	DOE 1988	DOE 1988	BEIR V	Appropriate approach	NA
G. INTRUSION SCENARIOS								

Parameter, assumption, or process modeled	G.1. GENERAL	G.1(a) Scenarios considered	G2. INTRUDER-DRILLING SCENARIO	G.2(a) Dust loading	G.2(b) Volume of waste excavated	G.2(c) Dilution factor	G.2(d) Duration of exposure
ALLW PA	Concrete PA (Maheras, et al. 1992)	Acute drilling, acute construction (excavation), chronic agricultural, and chronic radon		Wet drilling, no inhalation	0.023 m ³ or 0.21 m ³ based on 8-in or 22-in diameter wells; 1990 RWM C PA analysis, 11-cm diameter well	None used; the inventory was spread onto the ground surface, resulting in a plane source of 2,200 m ² , and doses were calculated with MICROSHIELD; this is a different scenario than that used for the IEB PA, where the cuttings were diluted with clean cuttings and put into a pit	1 hr of inhalation exposure, 40 hr of external exposure, based on a residential size well; one run evaluated effect of longer inhalation exposure
ALLW PA	Concrete PA (Maheras, et al. 1992)	Acute drilling, acute construction (excavation), chronic agricultural, and chronic radon		Wet drilling, no inhalation	0.023 m ³ based on 1990 RWM C PA analysis, 11-cm diameter well	0.013	6 hr to uncovered cuttings, 494 hr for covered cuttings
HBPA	(Rood 1992)	Acute drilling		Wet drilling, no inhalation	NA	NA	6 hr to uncovered cuttings, 494 hr for covered cuttings
IWPF PA	(Rood 1993)	None		NA	1.5 m ³ for a 22-in. diameter well in pits and 0.75 m ³ for a 22-in. diameter well in soil vaults	NA	160 hr inhalation and external exposure, the time it takes to drill and develop a 22-in. irrigation well
RWMCILW	(Maheras et al. 1994)	Acute drilling, acute construction, chronic agricultural, chronic radon		1x10 ⁻³ g/m ³	NA	For pits, 1.5 m ³ waste/2,200 m ² area x 1.75 m thickness = 9.1 x 10 ⁻⁴ ; for soil vaults, 0.75 m ³ /2,200 m ² area x 1.75 m thick = 4.5 x 10 ⁻⁴	NA
Preliminary Scoping Risk Assessment (Loehr et al. 1994)		None		NA	More conservative than others except for RWM C PA	ALLW conservative	ALLW non-conservative except for one inhalation exposure
Degree of conservatism in Spread	Spread, sensitivity, and importance	More conservative than some		Spread high, sensitivity high, importance 9/9	Spread high, sensitivity high, importance 9/9	Spread high, sensitivity moderate, importance 6/9	Spread high, sensitivity moderate, importance 6/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
G.2(e) Time of intrusion	100 yr after disposal facility closure for most runs; also explored times up to 1,000,000 yr	100 yr	100 yr	NA	The intruder may enter the site any time after institutional control; peak was not at 100 yr	NA	Consistent approach taken	NA
G.2(f) Code(s) used	GENII	MICROSHIELD	PATHRAE and MICROSHIELD	NA	MICROSHIELD and GENII	NA	Range of common codes used	NA
G.2(g) Other				NA	This analysis is similar to the concrete performance assessment and differs from the IEB PA; the major difference is in how the cuttings are dealt with; in the IEB PA, a mud pit was assumed; in later analysis, it was found that drillers in the local area do not use mud pits, but spread the cuttings out over the area	NA	NA	NA
G.3. INTRUDER-EXCAVATION SCENARIO		The intruder excavation scenario was not evaluated because the intruder could not penetrate the waste; therefore, no doses from dust inhalation could be calculated						
G.3(a) Dust loading	$5.0 \times 10^{-3} \text{ g/m}^3$	NA	NA	NA	$1 \times 10^{-3} \text{ g/m}^3$	NA	More conservative than RWMC PA	Spread medium, sensitivity high, importance 6/9
G.3(b) Volume of waste excavated	100 m ³ (waste and backfill) of which 75 m ³ are waste	NA	NA	NA	60 m ³ (at the time of maximum erosion, 2.4 m of cover remain; therefore, 0.6 m of waste is brought to the surface)	NA	Different scenarios considered	Spread medium, sensitivity high, importance 6/9

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
G.3(c) Dilution factor	$(75 \text{ m}^3 \text{ waste}) / (100 \text{ m}^3 \text{ waste and backfill} + 152 \text{ m}^3 \text{ soil}) = 0.298$	NA	NA	NA	This factor was not used; the intruder is exposed to contaminated dust during basement excavation and external exposure from the waste exposed at the base of the basement	NA	NA	NA
G.3(d) Duration of exposure	80 hr	NA	NA	NA	64 hr	NA	ALLW more conservative than RWMC PA	Spread small, sensitivity medium, importance 2/9
G.3(e) Time of intrusion	500 yr after disposal facility closure for most runs; also explored times up to 1,000,000 yr	NA	NA	NA	Maximum doses were predicted to occur at 10^6 yr from U-238 decay chain; protective cover assumed to last 3,000 yr	NA	Consistent approach taken	Spread high, sensitivity high, importance 9/9
G.3(f) Code(s) used	GENII	NA	NA	NA	MICROSHIELD for external exposure; GENII for inhalation doses	NA	Appropriate codes used	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
G.4. INTRUDER- AGRICULTURAL SCENARIO								
G.4(a) Scenario description	Intrusion was assumed to occur at 100 yr (drilling) and 500 yr (excavation); drill cuttings or excavation materials are mixed with soil used for crops; crops are consumed by farmer and livestock; farmer consumes water, contaminated crops, and animal products; intrusion times up to 1,000,000 yr were explored	Contaminated drill cuttings were spread over the intrusion site; no waste from basement excavation was considered	Contaminated cuttings are spread over the intrusion site; no waste from basement excavation was considered	NA	The intruder drills a well and spreads cuttings over a 2,200-m ² area and mixed in soil to a depth of 0.75 m where crops are grown; in addition, a home is constructed, and the waste brought to the surface during basement excavation is also spread out on the agriculture field	NA	ALLW PA more conservative than all except RWMC PA	Spread high, sensitivity high, importance 9/9
G.4(b) Dust loading	1.0×10^{-4} g/m ³	5.53×10^{-5} g/m ³ (weighted average)	5.53×10^{-5} g/m ³ (weighted average)	NA	5×10^{-5} g/m ³ ; this is a weighted average for different field activities	NA	ALLW conservative	Spread low sensitivity high, importance 3/9
G.4(c) Volume of waste excavated	0.34 m ³ for drilling-agricultural scenario (100 yr) and 100 m ³ for excavation-agricultural scenario	0.028 m ³ (8-in. well) and 0.21 m ³ (22-in. well)	0.023 m ³	NA	0.2 m ³ from well drilling and 60 m ³ from basement excavation	NA	ALLW PA more conservative than all except RWMC PA	Spread high, sensitivity high, importance 9/9
G.4(d) Dilution factor	$0.34/(375 + 0.34) = 9.06 \times 10^{-4}$	6.1×10^{-4}	1.15×10^{-5}	NA	The waste was spread out on a 2,200-m ² lot and mixed to a depth of 0.75 m; therefore, the dilution factor is 0.036	NA	Wide range of values used; ALLW PA in middle of range	Spread high, sensitivity high, importance 9/9
G.4(e) Groundwater included (y/n)	No	No	No	NA	Yes, used all pathway doses at selected times	NA	ALLW PA same as most others but less conservative than RWMC PA	Spread medium, sensitivity medium, importance 6/9
G.4(f) Code(s) used	GENII was used to calculate doses from ingestion, inhalation, and external exposure	GENII was used to calculate ingestion doses; MICROSHIELD was used to calculate external exposure	PATHRAE, MICROSHIELD	NA	GENII for inhalation and ingestion, MICROSHIELD for external exposure	NA	Appropriate codes used	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
G.4(g) Other		The intruder agriculture dose was based on two different diets, one from Yang and Nelson and the other from Rupp; two different inhalation rates were also considered					NA	NA
G.5. RADON EXPOSURE SCENARIO		None	None	NA		NA		
G.5(a) Diffusion coefficient	$2.5 \times 10^{-4} \text{ m}^2/\text{s}$ in cover $2.5 \times 10^{-8} \text{ m}^2/\text{s}$ in foundation	NA	NA	NA	$2.5 \times 10^{-8} \text{ m}^2/\text{s}$ in foundation	NA	NA	NA
G.5(b) Emanation coefficient	0.17	NA	NA	NA	0.17	NA	Typical value used	Spread low, sensitivity medium, importance 2/9
G.5(c) Distance from bottom of basement to top of waste	0.3 m	NA	NA	NA	0 m for pits; 0.3 m for soil vaults	NA	ALLW less conservative than RWMC PA	Spread moderate, sensitivity moderate, importance 4/9
G.5(d) Dose conversion method	Methodology used in RESRAD; used definition of working levels per Radiological Health Handbook and multiplied by exposure time for dose to receptor; overall dose conversion was $1\text{pCi/L} = 130 \text{ mrem/yr}$	NA	NA	NA	Methodology used in RESRAD; used definition of working levels per Radiological Health Handbook and multiplied by exposure time for dose to receptor; overall dose conversion was $1\text{pCi/L} = 130 \text{ mrem/yr}$	NA	Methods not comparable especially since ALLW did not calculate dose	Spread large, sensitivity moderate, importance 6/9
G.5(e) Code(s) used	RESRAD	NA	NA	NA	RESRAD	NA	NA	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
H. REGULATORY STANDARDS USED TO COMPARE CALCULATED DOSES	Performance was measured against DOE Order 5820.2A for all-pathway analysis and 40 CFR 141 (or DOE Order 5400.5) for drinking water standards	Performance was measured against DOE Order 5820.2A for all-pathway analysis and 40 CFR 141 (or DOE Order 5400.5) for drinking water standards	Performance was measured against DOE Order 5820.2A for all-pathway analysis and 40 CFR 141 (or DOE Order 5400.5) for drinking water standards	Performance was measured against DOE Order 5820.2A for all-pathway analysis and 40 CFR 141 (or DOE Order 5400.5) for drinking water standards	Performance was measured against DOE Order 5820.2A for all-pathway analysis and 40 CFR 141 (or DOE Order 5400.5) for drinking water standards	Carcinogenic risks are compared to the National Contingency Plan target range of 10^{-4} to 10^{-6} ; noncarcinogenic risks are evaluated using the Hazard Index (HI); if the HI for a contaminant is >1 , then there may be concern for potential adverse health effects	Performance was compared against appropriate regulatory standards	NA
I. RESULTS		Pu-239, and Am-241 were the primary nuclides in the intruder analysis, max dose=from 64 to 82 mrem/yr (intruder agriculture) at 100 yr; Cs-137 contributed about 4% in the form of external exposure	Maximum intruder dose was for Am-241 (4.9 mrem) for the agriculture scenario at 100 yr					

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
I.1. UNDISTURBED- WASTE SCENARIOS								
I.1(a) Dominant pathways	Vadose zone → groundwater (migration); ingestion (exposure)	Groundwater	Groundwater for dose at time periods greater than 10,000 yr and intruder agriculture for time periods less than 10,000 yr	All pathway	Groundwater, all pathway scenario; max. dose after institutional control at 100 m south of the RWMC boundary; max. all pathway dose 17 mrem	Ingestion of crops and groundwater are most important pathways	NA	NA
I.1(b) Dominant radionuclides	Am-241 decay chain	U-238 and Am-241; maximum dose of 150 mrem/yr at 7.4×10^5 yr (driven primarily by daughter ingrowth)	U-238 and Am-241 (Np-237 progeny the primary dose contributor); maximum dose of 420 mrem/yr at 7.2×10^5 yr (driven primarily by daughter ingrowth)	U-233 and U-235; dose of 0.2 mrem (U- 233) and 0.6 mrem (U-235); Np-237 was the only nuclide to arrive at the receptor before 10,000 yr	C-14, H-3 and I-129	All pathways: Sr-90 risk $>10^{-4}$; Am-241, Cs-137, Pu-239, Pu- 240, Ra-226, U-235 risks are between 10^{-4} and 10^{-6} ; groundwater: Am-241, C-14, Ni-59, Np-237, U-234, and U- 238 risks are between 10^{-4} and 10^{-6} ; U-238 HI is 0.1 to 1.0; nonradiological: risk $<10^{-6}$ and HI <1.0	NA	NA
I.1(c) Important parameters	Radionuclide travel time in the vadose zone (effective K_d values), TRU activity loading, degradation of FWF, distance to receptor well	K_d value in soils and unsaturated zone; this study indicates that waste containment does very little to improve the performance for long-lived actinides; the sorption coefficient in the unsaturated zone is perhaps the most important parameter in addition to the unsaturated zone flow characteristics that determine the travel time of the nuclides	K_d value in soils and unsaturated zone; this study indicates that waste containment does very little to improve the performance for long-lived actinides; the sorption coefficient in the unsaturated zone is perhaps the most important parameter in addition to the unsaturated zone flow characteristics that determine the travel time of the nuclides	K_d value and the infiltration rate	Leach rates of C-14 from miscellaneous waste and activated metals	Release rate parameters, cover thickness, infiltration rate, mass of unknown radionuclides and chemicals, synergistic effects of contaminants, and land use scenarios	NA	NA

Parameter, assumption, or process modeled	ALLW PA	Concrete PA (Maheras, et al. 1992)	IEB PA (Rood 1992)	IWPF PA (Rood 1993)	RWMC LLW Radiological PA (Maheras et al. 1994)	Preliminary Scoping Risk Assessment (Loehr et al. 1994)	Degree of conservatism in ALLW PA	Spread, sensitivity, and importance
I.2. INTRUDER SCENARIOS								
I.2(a) Dominant pathways	Inhalation for both acute and chronic scenarios	Intruder agriculture scenario, inhalation (Am-241) groundwater ingestion (U-238); for external exposure, Cs-137 was the dominant nuclide	Intruder agriculture, inhalation (Am-241) and external exposure (Cs-137)	NA	External exposure in the soil vault chronic intruder-agriculture scenario was primary pathway; second most important pathway had equal contributions from inhalation and exposure during the soil vault scenario	None considered	NA	NA
I.2(b) Dominant radionuclides	Cs-137 (acute), Cs-137 and Am-241 (chronic)	Am-241 and U-238	Am-241 and Cs-137 (see above)	NA	Cs-137 for dominant pathway; Cs-137 and Ni-63 for second most important pathway; U-238 decay chain for waste in pits	NA	NA	NA
I.2(c) Important parameters	Dust loading factor, TRU activity loading, time of human intrusion	Well diameter, assumed diet and inhalation rate	Dissolution rate of IEB waste form, and K_d values for actinides	NA	Exposure time for dominant pathway; sensitivity analysis performed only for dominant pathway	NA	NA	NA
I.2(d) Other					If leaching (source depletion) had been considered, it would have affected only the maximum doses for the pits and not the soil vaults; the maximums for the soil vaults occurred at the end of institutional control, while those for the pits occurred at 1×10^6 yr	NA	NA	NA