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Performance Evaluation of the Technical Capabilities of DOE Sites for Disposal of Mixed Low-Level Waste

Volume 2: Technical Basis and Discussion of Results

Prepared for the Department of Energy (DOE)
Office of Waste Management
Federal Facility Compliance Act
Disposal Workgroup

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March 1996
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ABSTRACT

A team of analysts designed and conducted a performance evaluation to estimate the technical capabilities of fifteen Department of Energy sites for disposal of mixed low-level waste (i.e., waste that contains both low-level radioactive materials and hazardous constituents). Volume 1 summarizes the process for selecting the fifteen sites, the methodology used in the evaluation, and the conclusions derived from the evaluation. Volume 2 provides details about the site-selection process, the performance-evaluation methodology, and the overall results of the analysis. Volume 3 contains detailed evaluations of the fifteen sites and discussions of the results for each site.

PREFACE

This report documents the performance evaluation of facilities at various Department of Energy (DOE) sites relative to their capabilities for the disposal of mixed low-level waste (MLLW). The principal goal in developing the performance evaluation (PE) was to estimate the limiting concentrations of radionuclides in residuals resulting from treatment of MLLW for disposal at these sites. The report consists of three volumes:

Volume 1 is an executive summary both of the PE methodology and of the results obtained from the PEs. While this volume briefly reviews the scope and method of analyses, its main objective is to emphasize the important insights and conclusions derived from the conduct of the PEs.

Volume 2 first describes the screening process used to determine the sites to be considered in the PEs. This volume then provides the technical details of the methodology for conducting the performance evaluations. It also provides a comparison and analysis of the overall results for all sites that were evaluated.

Volume 3 presents the results of the PEs for the 15 sites considered in the process. This presentation includes a discussion of the conceptual models and data used in the PE for each site.

The PE is not a substitute for the detailed analyses provided by performance assessments required by DOE Order 5820.2A; rather, it is a means for the DOE and the States to begin evaluating options for disposal of MLLW treatment residuals. The ultimate identification of sites that may host MLLW disposal activities will follow state and federal regulations for siting and permitting and will include public involvement in the decision-making process. The appropriate site-specific performance or risk assessments and environmental impact analyses in accordance with the National Environmental Policy Act will be required in determining limits on quantities of radionuclides that may be acceptable for disposal at any site.

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NOMENCLATURE

A	Plan area of disposal facility (m^2)
a_1	Length of disposal facility parallel to groundwater flow (m)
a_2	Width of disposal facility perpendicular to groundwater flow (m)
A_D	Atmospheric dispersion term ([$\mu Ci/m^3$]/[μ Ci/m^2-s])
ANLE	Argonne National Laboratory—East
C_{Atm}	Radionuclide-specific permissible concentration in air ($\mu Ci/m^3$)
CRF	Concentration reduction factor (dimensionless)
CRF_{Atm}	Concentration reduction factor for atmospheric releases (dimensionless)
CRF_{Diff}	Concentration reduction factor for environmental transport by soil diffusion in atmospheric pathway (dimensionless)
CRF_{Disp}	Concentration reduction factor for environmental transport by dispersion in air in atmospheric pathway (dimensionless)
CRF_{Source}	Concentration reduction factor for source in water pathway (dimensionless)
CRF_{Water}	Concentration reduction factor for environmental transport in groundwater or surface water in water pathway (dimensionless)
CSTP	Conceptual site treatment plan
C_{Atm}	Radionuclide concentration in ambient air resulting from the air concentration at the soil surface ($\mu Ci/m^3$)
$C_{Leachate}$	Radionuclide-specific concentration in leachate exiting the disposal facility ($\mu Ci/L$)
C_{SS}	Radionuclide concentration at the soil surface above disposal facility in atmospheric pathway ($\mu Ci/m^3$)
C_W	Permissible waste concentration for a radionuclide; the smallest of $C_{W\text{-}Water}$, $C_{W\text{-}Atm}$, and $C_{W\text{-}Intr}$ ($\mu Ci/m^3$)
$C_{W\text{-}Atm}$	Radionuclide-specific waste concentration for permissible atmospheric releases ($\mu Ci/m^3$)
$C_{W\text{-}Intr}$	Radionuclide-specific waste concentration for permissible exposures of an inadvertent intruder ($\mu Ci/m^3$)
$C_{W\text{-}Water}$	Radionuclide-specific waste concentration for permissible releases to groundwater or surface water ($\mu Ci/m^3$)
C_{Waste}	Radionuclide-specific concentration in waste ($\mu Ci/L$)
C_{Water}	Radionuclide-specific permissible concentration in drinking water ($\mu Ci/L$)
d	Depth of surface soil (m)
D	Diffusion coefficient in air (m^2/s)
DCF_{Ing}	Radionuclide-specific dose conversion factor for ingestion (rem/μCi)
D_l	Radionuclide-specific moisture diffusion coefficient (m^2/s)
d_m	Mixing depth in groundwater (m)
DOE	Department of Energy
DSTP	Draft site treatment plan
DWG	Disposal Work Group
EM PEIS	Environmental Management Programmatic Environmental Impact Statement
EPA	Environmental Protection Agency
EPRI	Electric Power Research Institute
FEMP	Fernald Environmental Management Project
FFCAct	Federal Facility Compliance Act
f_{et}	Fraction of year exposed
f_m	Fraction of volume of disposal facility containing waste at time of facility closure, taking into account the presence of uncontaminated material in the facility (dimensionless)
f_s	Shielding factor of home during indoor exposure
HAN	Hanford Reservation
H_{Atm}	Performance measure of 10 mrem (0.10 mSv)/y effective dose equivalent for atmospheric releases
H_{Intr}	Performance measure of 100 mrem (1 mSv)/y effective dose equivalent for exposure of inadvertent intruders
$H_{Intr(x)}$	Total dose to inadvertent intruders for scenario x (rem/y)
HLW	High-level waste
H_{Water}	Performance measure of 4 mrem (0.04 mSv)/y effective dose equivalent for the water pathway
I	Natural recharge of water through local soils at disposal site (cm/y)
ICRP	International Commission on Radiological Protection
INEL	Idaho National Engineering Laboratory
IR_a	Inhalation rate of air (m^3/y)
IR_s	Soil consumption (intake) rate (kg/y)
IR_v	Vegetable consumption (intake) rate (kg fresh weight/y)
IR_{Water}	Intake rate for ingestion of drinking water (L/y)
J	Radionuclide flux density through the soil above the waste disposal facility ($\mu Ci/m^2$-s)
KAPL-K	Knolls Atomic Power Laboratory—Kesselring
K_d^G	Radionuclide-specific distribution coefficient for grout (mL/g)
K_d	Radionuclide-specific distribution coefficient for the geologic media (mL/g)
l	Vertical distance between disposal facility and groundwater (m)
L_a	Atmospheric mass loading of surface soil (kg/m^3)

LANL	Los Alamos National Laboratory
LDR	Land Disposal Restriction
LLNL	Lawrence Livermore National Laboratory
LLW	Low-level waste
MLLW	Mixed low-level waste
MTRU	Mixed transuranic waste
MWIR	Mixed Waste Inventory Report
n	Porosity of saturated zone
NCRP	National Council on Radiological Protection and Measurements
NEPA	National Environmental Policy Act
NGA	National Governors' Association
NRC	Nuclear Regulatory Commission
NTS	Nevada Test Site
ORR	Oak Ridge Reservation
PA	Performance assessment
PAGAN	One-dimensional flow, three-dimensional transport computer code
$PDCF_{Water}$	Radionuclide-specific pathway dose conversion factor for ingestion of drinking water ([rem/y]/[μCi/L])
PE	Performance evaluation
PGDP	Paducah Gaseous Diffusion Plant
PORTS	Portsmouth Gaseous Diffusion Plant
PSTP	Proposed site treatment plan
QA	Quality assurance
QA/QC	Quality assurance/quality control
Q_f	Volumetric flow of leachate from disposal facility (m^3/y)
q_f	Rate of water flowing through disposal facility (m/y)
Q_{gw}	Volumetric groundwater flow (m^3/y)
q_{gw}	Groundwater Darcy velocity (m/y)
r	Ratio of density of water in air to that in liquid phase for H-3 (dimensionless), or ratio of CO_2 concentration in air to that dissolved in water for ^{14}C (dimensionless)
RCRA	Resource Conservation and Recovery Act
RFETS	Rocky Flats Environmental Technology Site
r_{Decay}	Radionuclide-specific term accounting for radioactive decay between time of disposal and estimated time of exposure (dimensionless)
s	Percent saturation of the geologic media
$SDCF_{Atm}$	Radionuclide-specific scenario dose conversion factor for all exposure pathways for atmospheric releases ([rem/y]/[μCi/m^3])
$SDCF_{Intr}$	Radionuclide-specific scenario dose conversion factor for all exposure pathways for inadvertent intruders ([rem/y]/[μCi/m^3])
SNL	Sandia National Laboratories
SRS	Savannah River Site
STP	Site treatment plan
SWSA	Solid Waste Storage Area
t_a	Radionuclide arrival time at performance boundary (y)
t_{cs}	Retarded contaminant travel time in saturated zone (y)
t_{cv}	Retarded contaminant travel time in vadose zone (y)
t_d	Diffusion transport time from the waste to the surface (y)
t_i	Time of intrusion for the scenario being analyzed (y)
t_{ws}	Water travel time in saturated zone (y)
t_{wv}	Water travel time in vadose zone (y)
$t_{1/2}$	Radionuclide half-life (y)
v_d	Diffusion velocity (m/s)
WVDP	West Valley Demonstration Project
x	Cover thickness above disposal facility (m)
ρ_b	Dry bulk density of the geologic media in vadose or saturated zone (g/cm^3)
θ_G	Volumetric water content of grouted waste form (mL/mL [i.e., dimensionless])
ρ_G	Dry bulk density of grouted waste form (g/cm^3)
θ_W	Volumetric moisture content of vadose zone (mL/cm^3)

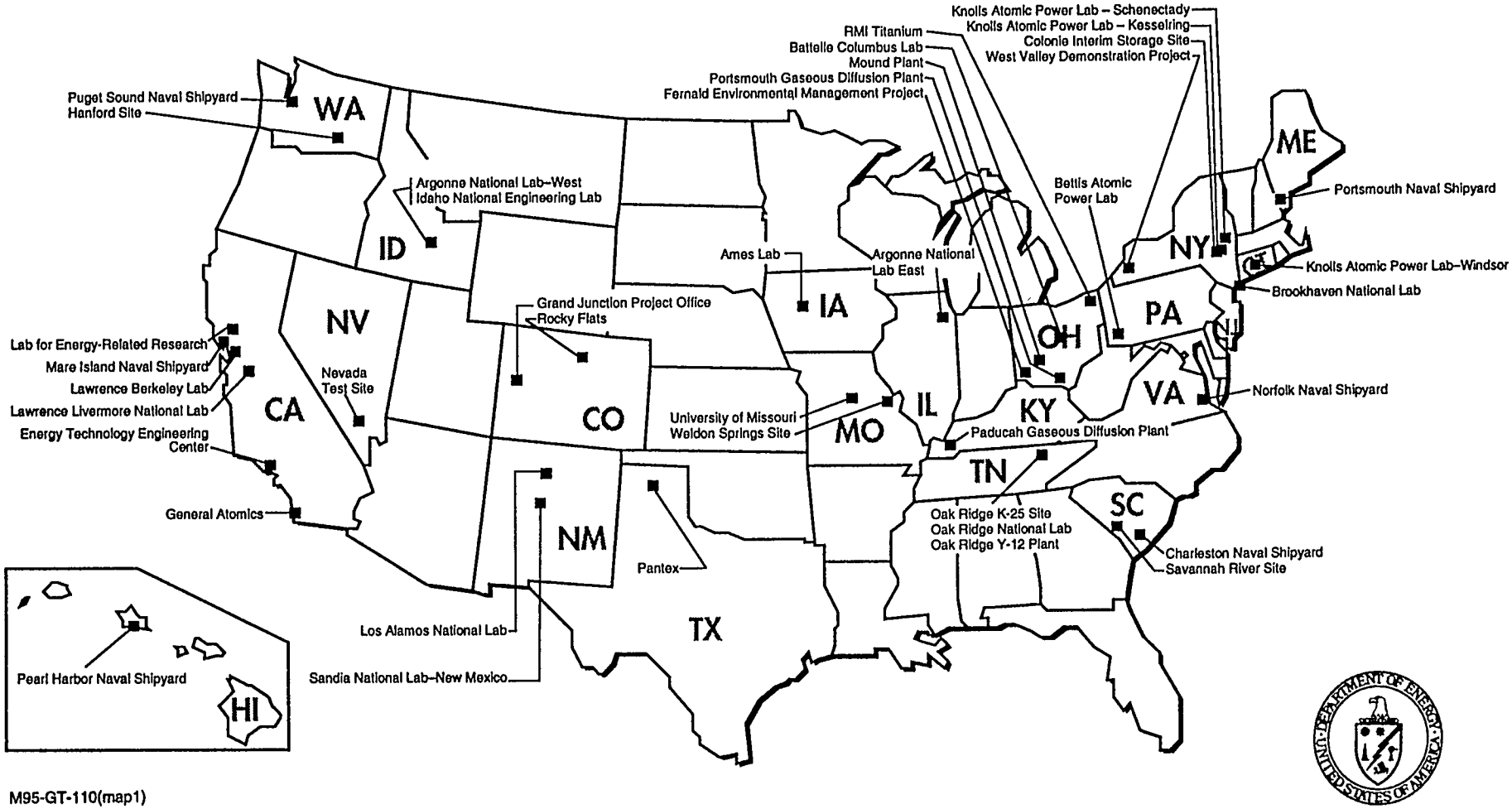
1. INTRODUCTION

For more than 50 years, the United States has produced materials for nuclear weapons and has conducted research with nuclear materials. These activities generated wastes that contain both radioactive and hazardous (toxic chemical) components; these are called mixed waste. The Department of Energy (DOE) is faced with the challenge of managing these wastes. The DOE currently generates, stores, or is expected to generate over the next five years 650,000 m^3 of mixed waste at 41 sites in 20 states (Figure 1-1). About 72% of this waste can be categorized as high-level waste (HLW), 20% is mixed low-level waste (MLLW), and approximately 8% is mixed transuranic waste (MTRU).

Because it has a hazardous component, mixed waste must be treated to comply with Land Disposal Restrictions (LDRs) of the Resource Conservation and Recovery Act (RCRA, 1976). However, there is insufficient capacity, and in some cases a lack of available technologies, to treat these wastes. The Federal Facilities Compliance Act (FFCAct) of 1992 (FFCAct, 1992) requires the Secretary of Energy to develop and submit site treatment plans (STPs) for the development both of treatment capacity and of technologies for treating mixed waste for each facility at which the DOE stores or generates these wastes. These plans identify how the DOE will provide necessary mixed waste treatment capacity, including schedules for bringing new treatment facilities into operation. In collaboration with the States and the National Governors Association (NGA), the DOE has been evaluating candidate treatment options and developing the required treatment plans.

Because of the large scope of the task, the DOE has followed a three-phased approach for the development of STPs for the sites that store, generate, or expect to generate DOE mixed waste (the Hanford site is exempt from the requirement to prepare an STP because there is an agreement in place that meets the FFCAct requirements). In Phase I, the DOE developed conceptual site treatment plans (CSTPs) that identified a wide range of treatment options; the CSTPs were submitted to the appropriate regulatory agencies at the State and Federal level (U. S. Environmental Protection Agency [EPA]) in October 1993. In Phase II, the draft site treatment plans (DSTPs) were developed to narrow the treatment options to those that had been proposed. The DSTPs were submitted to the relevant regulatory agencies in August 1994. As part of the third phase, the DOE submitted in April 1995 the proposed site treatment plans (PSTPs) for approval. Implementation was formalized through consent orders issued by the regulatory agencies; the deadline of October 6, 1995 was met by most sites.

The types and volumes of the MLLW streams that will be treated are shown in Table 1-1. A summary of the proposed treatment options for MLLW and the associated volume of waste to be treated by each treatment type is provided in Table 1-2. All treatment methods will result in solid waste forms for disposal.



M95-GT-110(map1)

Figure 1-1. The 41 DOE installations currently storing or projecting to generate mixed waste.

Table 1-1. MLLW Waste Stream Type and Volume (based on the 1995 mixed waste inventory reported by the sites to DOE)

WASTE TYPE	VOLUME (m^3)	% of TOTAL VOLUME
Aqueous Liquid/Slurries	8600	6.8%
Organic Liquids	4600	3.6%
Residuals/Sludges/Salts	47,000	36.9%
Soils	13,900	10.9%
Lab Packs	900	0.7%
Debris	49,200	38.6%
Batteries/Lead/Mercury/Reactive Metals	2400	1.9%
Special Wastes/Other*	700	0.6%
TOTAL	**127,300**	**100.0%**

*Includes waste streams already meeting LDR standards

Table 1-2. Proposed Treatment Options for MLLW

WASTE TYPE	VOLUME OF WASTE (m^3)[a]	% OF TOTAL VOLUME
Alkali Metals Treatment	690	0.3%
Amalgamation	20	< 0.1%
Deactivation	810	0.4%
Inorganic Debris Treatment	8600	4.3%
Mercury Separation	1170	0.6%
Neutralization/Non-Aqueous	10	< 0.1%
None/Meets LDR	31,660	15.7%
Organic Destruction	24,280	12.1%
Pre- or Post-Treatment	26,620	13.2%
Soil Washing	6590	3.3%
Stabilization	57,270	28.5%
To Be Determined	29,250	14.5%
Wastewater Treatment	4430	2.2%
Waste Isolation Pilot Plant	9830	4.9%
TOTAL	**201,230**	**100.0%** [b]

a Includes both wastes with and without identified treatment options and waste stream volumes proposed for treatment by multiple systems (i.e., treatment train volumes). Consideration of multiple treatment systems causes total volume to be greater than the total in Table 1-1.

b Does not add to 100.0% because of rounding

Although the FFCAct does not specifically require the DOE to address disposal of treated mixed waste, both the DOE and the States realize that the method of treatment for a specific waste is an integral component of any considerations of methods for its disposal. As a result, the DOE established the FFCAct Disposal Workgroup (DWG) in June 1993 to work with the States to define and develop a process for evaluating disposal options. The focus of the DWG was to identify, from among the sites currently storing or expected to generate MLLW, sites that were suitable for further evaluation regarding their disposal capability. Some sites that had been determined to have marginal or no potential for disposal activities were removed or postponed from further evaluation under this process. Remaining sites were evaluated more extensively using a performance evaluation (PE) process that served as a preliminary scoping analysis. The PE was a technical analysis and did not address any ethical, social, or policy considerations relevant to siting MLLW disposal facilities. Further detail on the screening process for potential disposal sites is provided in Chapter 2 of this volume.

Application of the performance evaluation was limited to those sites with the potential for MLLW disposal as determined in the DWG screening process. Established processes are already being implemented for studying, designing, constructing, and ultimately operating disposal facilities for HLW and MTRU wastes (i.e., HLW repository and Waste Isolation Pilot Plant); therefore, disposal options were not considered in the PE process for either HLW or MTRU.

The DOE's Environmental Management Programmatic Environmental Impact Statement Project (EM PEIS Project) has analyzed potential costs, risks, transportation, and other environmental impacts of using each of the potential sites for some level of disposal activity. In conducting the PE, the PEIS data served as additional sources for use in discussions with personnel at the candidate sites to determine what analyses and data were appropriate for incorporation into the PE.

2. EVALUATION PROCESS FOR MLLW DISPOSAL OPTIONS

This chapter describes how the disposal planning process was established by the Disposal Workgroup, what major activities were accomplished, and how the performance evaluation results are expected to be used in the disposal configuration study.

2.1 SCREENING PROCESS FOR POTENTIAL DISPOSAL SITES

The process adopted by the DWG for evaluating disposal options is summarized in Figure 2-1. In October 1993, the DOE prepared a draft report describing the history and status of the DOE's low-level waste (LLW) and MLLW disposal program; the draft report also outlines a disposal planning process (DOE, 1994). The sites originally evaluated in this process were the 49 reported to Congress by the DOE in the Mixed Waste Inventory Report (MWIR) (April 1993) as currently storing or expected to generate MLLW* (Table 2-1). In the initial step of this process, sites that were in geographic proximity were combined into a single site for consideration in subsequent steps. Following this approach, the following sites were combined:

- Lawrence Livermore National Laboratory and Sandia National Laboratories (California)
- Idaho National Engineering Laboratory and Argonne National Laboratory (West)
- Sandia National Laboratories (New Mexico) and Inhalation Toxicology Research Institute
- Oak Ridge National Laboratory, Oak Ridge K-25 Site, and Oak Ridge Y-12 Site

This grouping reduced the number of sites to 44.

The DWG then established a screening process using exclusionary criteria for these 44 sites. These criteria were developed by reviewing Federal and State laws regarding the siting of waste treatment, storage, and disposal facilities to determine whether any criteria could be considered exclusionary minimum requirements for hosting disposal activities and could be applied uniformly across sites. At a joint DOE/States meeting in Tucson, Arizona, on March 3-4, 1994, an agreement was reached that in order to be further evaluated for potential disposal activities, a site

- must not be located within a 100-y floodplain,
- must not be located within 61 m of an active fault, and
- must have sufficient area to accommodate a 100-m buffer zone.

The first two criteria were derived from regulatory requirements under the Resource Conservation and Recovery Act (RCRA, 1976), which restricts the location of waste treatment, storage, and disposal facilities. The third criterion was derived from guidance from the EPA and the DOE concerning the minimum area required to properly operate a disposal facility.

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*Information compiled since 1993 indicates that the DOE currently generates, stores, or expects to generate (over the next five years) MLLW at 41 sites.

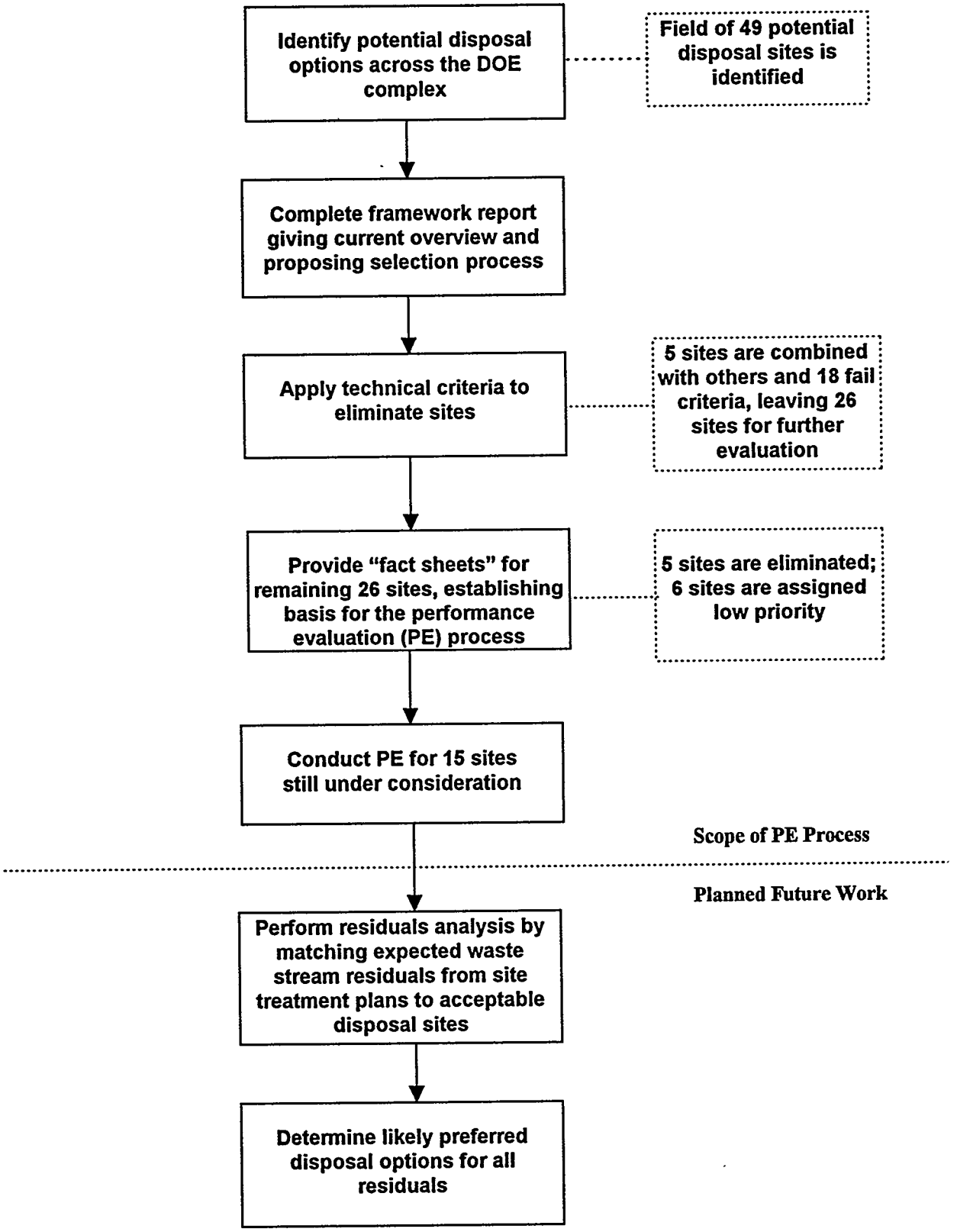


Figure 2-1. Evaluation process for DWG disposal options.

Table 2-1. The 49 DOE Installations Reported to Congress in 1993 as Storing or Generating MLLW (modified from DOE, 1994, Table 3-1)

CALIFORNIA
- Energy Technology Engineering Center
- General Atomics
- General Electric Vallecitos[a,b]
- Lawrence Livermore National Laboratory
- Lawrence Berkeley Laboratory
- Laboratory for Energy-Related Health Research
- Mare Island Naval Shipyard
- Sandia National Laboratories—California[b]

COLORADO
- Grand Junction Project Office
- Rocky Flats Plant

CONNECTICUT
- Knolls Atomic Power Laboratory, Windsor[a]

FLORIDA
- Pinellas Plant[b]

HAWAII
- Pearl Harbor Naval Shipyard

IDAHO
- Argonne National Laboratory—West
- Idaho National Engineering Laboratory

IOWA
- Ames Laboratory

ILLINOIS
- Argonne National Laboratory—East
- Site A/Plot M, Palos Forest Preserve[a,b]

KENTUCKY
- Paducah Gaseous Diffusion Plant

MAINE
- Portsmouth Naval Shipyard

MISSOURI
- Kansas City Plant[b]
- University of Missouri
- Weldon Springs Site Remedial Action Project

NEVADA
- Nevada Test Site

NEW JERSEY
- Middlesex Sampling Plant[b]
- Princeton Plasma Physics Laboratory[a,b]

NEW MEXICO
- Inhalation Toxicology Research Institute[b]
- Los Alamos National Laboratory
- Sandia National Laboratories—New Mexico

NEW YORK
- Brookhaven National Laboratory
- Colonie Interim Storage Site
- Knolls Atomic Power Laboratory, Kesselring
- Knolls Atomic Power Laboratory, Niskayuna
- West Valley Demonstration Project

OHIO
- Battelle Columbus Laboratory[a]
- Fernald Environmental Management Project
- Mound Plant
- Portsmouth Gaseous Diffusion Plant
- RMI Titanium, Inc.

PENNSYLVANIA
- Bettis Atomic Power Laboratory

SOUTH CAROLINA
- Charleston Naval Shipyard
- Savannah River Site

TENNESSEE
- K-25 Site, Oak Ridge Reservation
- Oak Ridge National Laboratory, Oak Ridge Reservation
- Y-12 Plant, Oak Ridge Reservation

TEXAS
- Pantex Plant

VIRGINIA
- Norfolk Naval Shipyard[a]

WASHINGTON
- Hanford Site
- Puget Sound Naval Shipyard

a At the time of the report to Congress, this installation was not currently producing or storing mixed waste but might do so in the next 5 years.
b No longer stores or plans to generate mixed waste

Application of the three exclusionary criteria identified 18 sites that did not meet the criteria. The results were presented at a March 30-31, 1994, joint DOE/States meeting in Dallas, Texas. At that time, an agreement was reached to remove the 18 sites from further evaluation and to prepare "fact sheets" on the remaining 26 sites. The fact sheets provided by the DOE would give additional site-specific information for identifying the strengths and weaknesses of the remaining sites for the purpose of disposal activities.

A predecisional draft of the report *Framework for DOE Mixed Low-Level Waste Disposal: Site Fact Sheets* (Gruebel et al., 1994) was subsequently prepared. On July 13, 1994, it was submitted for comment and review by the State representatives who had been discussing the selection of potential disposal sites. The DOE and the State representatives met on July 26-27, 1994, in Denver, Colorado, to discuss the fact sheet report and to consider proposals for elimination of sites from further evaluation. Prior to the meeting, the DWG had reviewed the information in the site fact sheets and evaluated the 26 sites according to the following methodology:

A. Factors contained in the fact sheets were grouped into three categories and evaluated:

   1. Technical Considerations—factors that represent the technical ability of a site to accommodate the waste disposal facility and minimize the risk of releases of waste constituents (e.g., precipitation and evapotranspiration, tectonic and volcanic hazard potential, soil stability and topography, flooding potential, groundwater hydrology [depth to groundwater]).

   2. Potential Receptor Considerations—factors that gauge the potential magnitude of consequences in the event a disposal facility ceases to function properly (e.g., population changes, significant groundwater resources, sensitive environment).

   3. Practical Considerations—factors that can potentially affect the development and long-term management of a disposal facility (ownership, mission, MLLW storage and generation, regulatory considerations).

B. The DWG evaluated each of the 26 sites according to the three categories. One of three results was assigned for each category: the site posed (1) a major problem, (2) a moderate problem, or (3) a minor problem. Sites with major problems were defined as having features or attributes that make developing and operating a disposal facility extraordinarily difficult. Moderate problems were defined as significant problems that could likely be solved with additional efforts and resources. Sites designated as having minor problems were those having neither major nor moderate problems.

Based on the DWG evaluation, during the July 1994 meeting the DOE proposed to the States that nine additional sites be eliminated from further consideration. Following the ensuing discussion, the DOE and the States agreed that the following sites would be eliminated from further evaluation:

Site	State
Energy Technology Engineering Center	California
General Atomics	California
General Electric Vallecitos Nuclear Center	California
Pinellas Plant	Florida
Site A/Plot M	Illinois

The DOE and the States also agreed that the following sites, while not eliminated from further evaluation, would be given a lower priority for further evaluation:

Site	State
Weldon Spring Remedial Action Project	Missouri
Brookhaven National Laboratory	New York
Mound Plant	Ohio
Bettis Atomic Power Laboratory	Pennsylvania
Knolls Atomic Power Laboratory—Niskayuna	New York

Sites in this group had issues that required further consideration. Such considerations included lack of data or information on the characterization of the technical attributes of the site, the volume of MLLW that may be generated by the site, and the existence and adequacy of other arrangements for disposal of the sites' MLLW. The DOE and the States agreed to further evaluate these sites in terms of their capabilities for on-site disposal of their own MLLW only if no other options could be identified through the disposal evaluation process. In no instance would these sites be considered as a disposal option for wastes from other sites; also, it was agreed these sites could be eliminated from further analysis should sufficient information suggest that their potential for disposal activities is too limited.

While the West Valley Demostration Project (WVDP) site remained on the list of potential disposal sites, DOE's roles and responsibilities for the WVDP are specifically delineated by the WVDP Act of 1980. Because the WVDP Act may prohibit disposal of waste from other sites, the DOE and the States agreed to evaluate this site for on-site radionuclides of concern.

In a subsequent decision, the DOE also assigned a lower priority for further evaluation to Knolls Atomic Power Laboratory—Kesselring (KAPL-K). Because of its exclusive mission as a research and development facility for the Department of the Navy, permanent disposal capacity for mixed waste is not part of the strategic planning for the site. The decision was also based on the very small volumes of its mixed waste streams (1.8 m^3 in storage, and waste generation projected at approximately 45 m^3 over the next five years), the desire to minimize shipments, and the fact that no onsite treatment is planned for KAPL-K's mixed waste streams. Similar to other low-priority sites, KAPL-K will be considered for development of mixed waste disposal capacity only in the event that no other viable option for its waste residues is found through disposal configuration analysis.

2.2 EVALUATION PROCESS FOR POTENTIAL DISPOSAL SITES

For the 15 sites not eliminated from further evaluation or assigned a lower priority for evaluation, a more technically detailed performance evaluation was conducted to increase

understanding of the strengths and weaknesses of a site's potential for a disposal facility and to better identify what types of disposal activities could or could not occur at a site. The sites considered in this analysis were the following (see also Figure 2-2):

Site	State
Lawrence Livermore National Laboratory, Site 300	California
Rocky Flats Environmental Technology Site	Colorado
Idaho National Engineering Laboratory	Idaho
Argonne National Laboratory-East	Illinois
Paducah Gaseous Diffusion Plant	Kentucky
Nevada Test Site	Nevada
Los Alamos National Laboratory	New Mexico
Sandia National Laboratories/New Mexico	New Mexico
West Valley Demonstration Project	New York
Fernald Environmental Management Project	Ohio
Portsmouth Gaseous Diffusion Plant	Ohio
Savannah River Site	South Carolina
Oak Ridge Reservation	Tennessee
Pantex Plant	Texas
Hanford Site	Washington

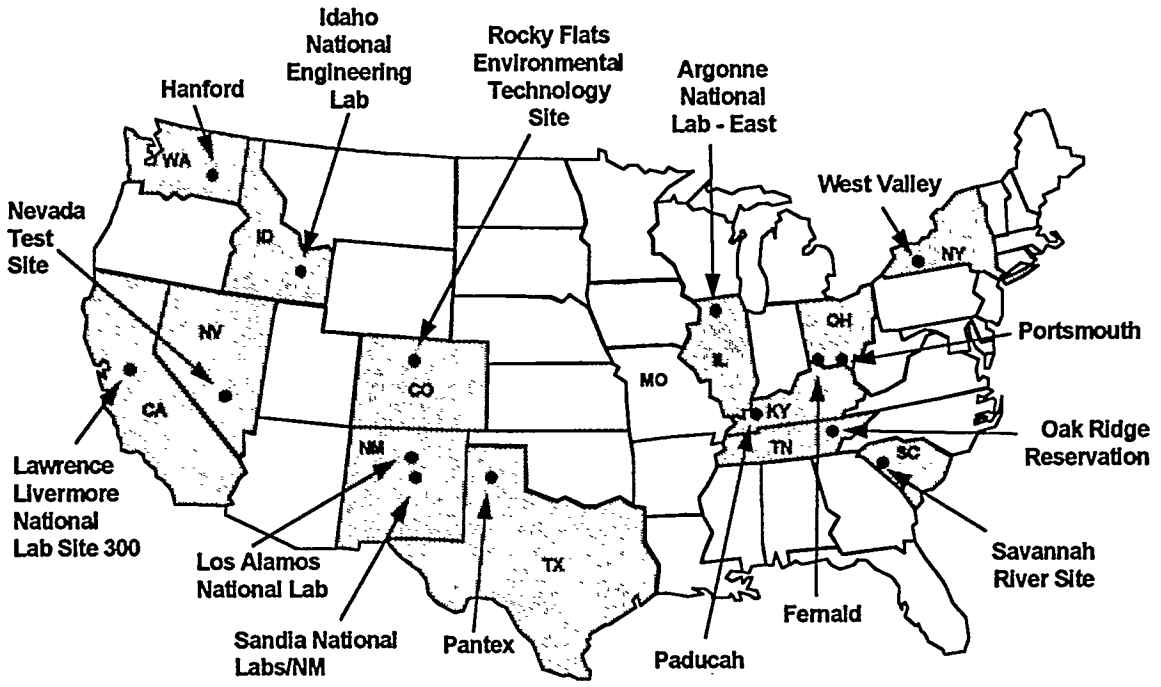


Figure 2-2. Sites considered in the PE.

The goal of the performance evaluation analysis was to quantify and compare the limitations of these 15 DOE sites for the disposal of MLLW. The objective was, therefore, to use a set of modeling assumptions of sufficient detail to capture major site-specific characteristics and yet be general enough for consistent application at all sites. The PE did not address any ethical, social, or policy considerations relevant to siting such facilities.

The results of the PE are presented in terms of concentrations of radionuclides in MLLW streams that a disposal facility at a site can accept without violating the general performance objectives prescribed in DOE Order 5820.2A (DOE, 1988a). This information will be used to identify the types of waste technically suitable for disposal at a given site.

2.3 DISPOSAL CONFIGURATION STUDY

In order to make an informed decision about the disposal of MLLW at each of the 15 sites (see Figure 2-3), certain activities will need to be conducted after the performance evaluations have been completed. These activities encompass the three steps outlined below. To gain stakeholder input and to resolve issues at the earliest possible stage, each step in the process will be coordinated with the States, and input from key stakeholders will be encouraged.

- *Develop Estimates of Waste Volumes and Radionuclide Concentrations in Treated MLLW Residues and Compare Concentrations to PE-Derived Radionuclide Concentrations*

  Once treatment methods are finalized through the FFCAct process, estimates of treated residue volumes and radionuclide concentrations will be developed for each of the MLLW streams. Waste streams that can be disposed of on-site will be identified, while other waste streams will be matched with disposal facilities at other sites that are technically capable of accepting them. Also, wastes that are eligible for disposal at a commercial facility will be identified at this step.

- *Develop On-Site Versus Off-Site Disposal Criteria*

  Even if the waste streams are technically acceptable for on-site disposal, a range of other factors will be considered in developing the criteria for deciding whether disposal of wastes can occur on-site. These factors include cost, facility capacity, compatibility of wastes, equity, transportation, and related issues. Input from the States and stakeholders will be critical to this step in the process.

- *Analyze Options of DOE's Disposal Configuration for MLLW*

  As a third step, complex-wide configurations for the disposal of treated MLLW will be developed; each such configuration will include the consideration of a full range of options. For purposes of the conduct of the performance evaluations, a disposal configuration is defined as a determination of the locations and sizes of a suite of disposal facilities. For each such suite, the sources and nature of the waste streams to be considered will be carefully designated, and each configuration option will be analyzed taking into consideration the relevant technical and non-technical issues. Again, input from the States and stakeholders is considered to be critical during this step.

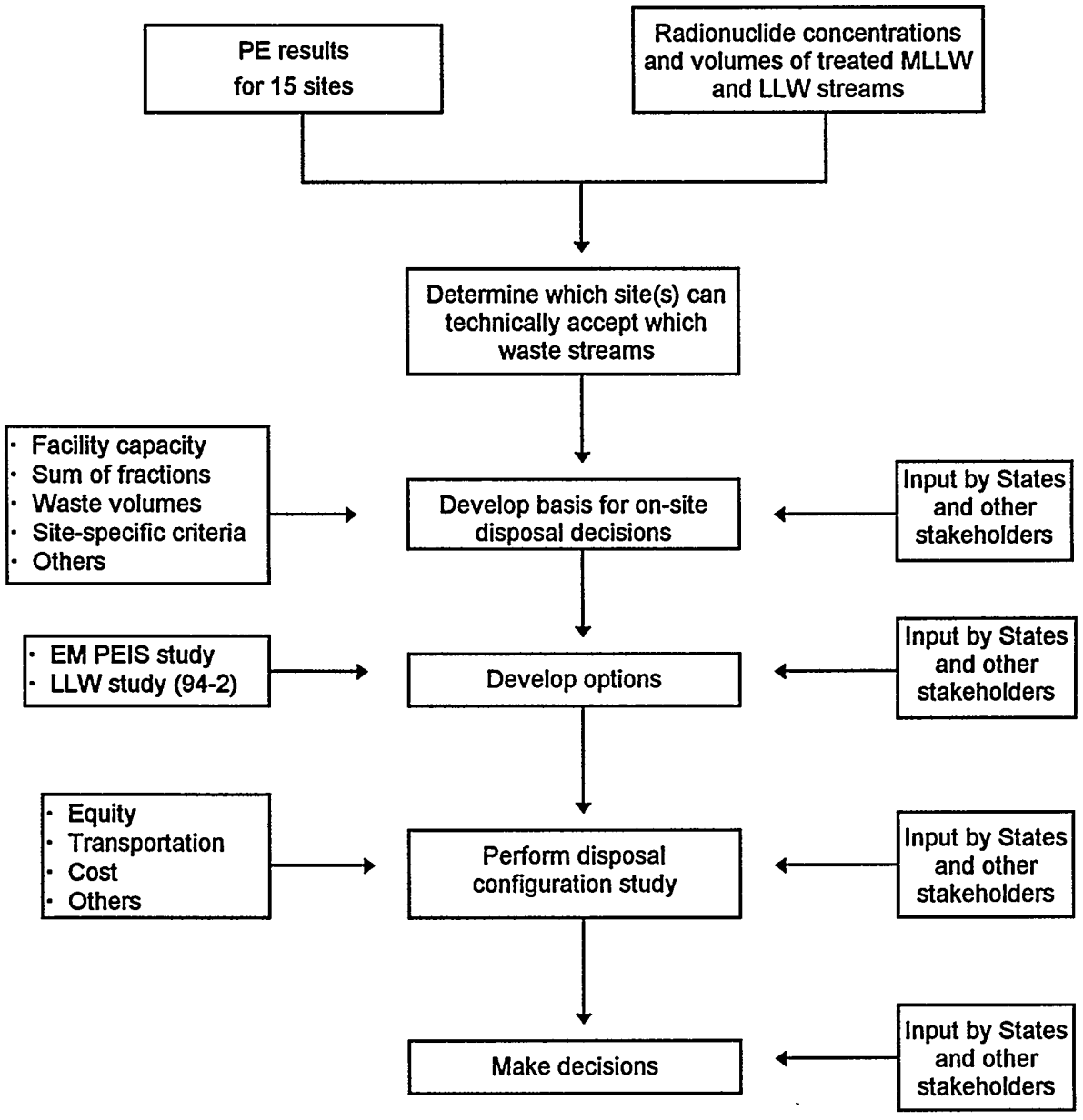


Figure 2-3. Disposal configuration study for MLLW.

3. IMPLEMENTATION OF PERFORMANCE EVALUATIONS

This chapter describes the organization and responsibilities for managing and performing the activities associated with the performance evaluations. The chapter also describes the quality control and quality assurance used in the PE analysis.

3.1 TEAMS

As shown in Figure 3-1, several technical teams implemented the PE and ensured the quality of the analysis. Members of these various groups are listed in Appendix A.

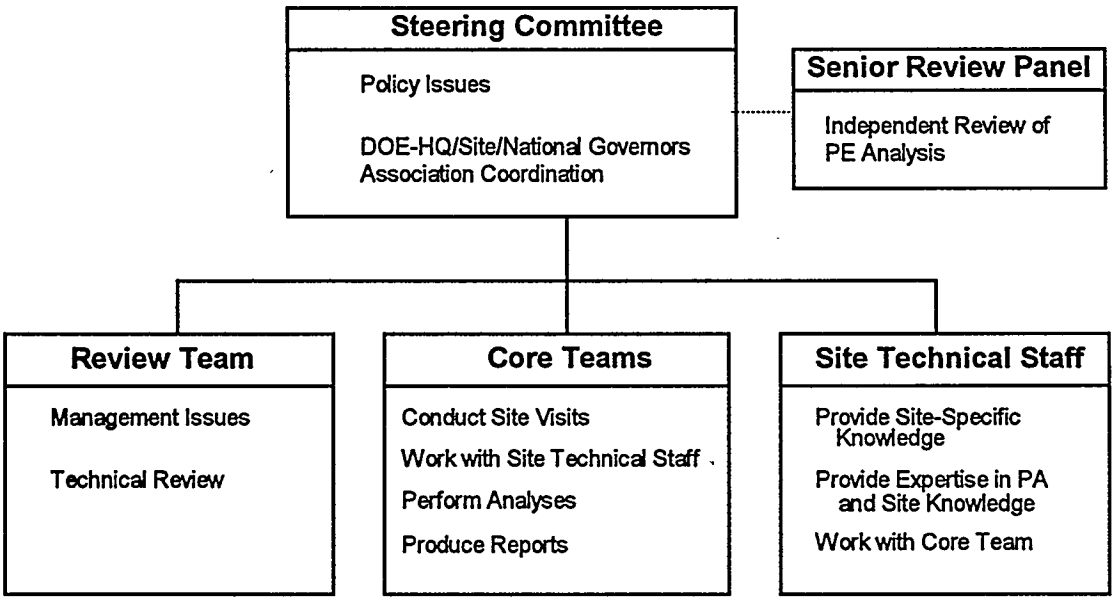


Figure 3-1. Roles and responsibilities of PE teams.

The Steering Committee consisted of four persons, two from DOE Headquarters and one each from the DOE Idaho Operations Office and Oak Ridge Reservation, who were familiar with both technical and policy-related MLLW issues. They provided coordination among the DOE/Headquarters (DOE/HQ), DOE sites, and the States, and presented policy-related issues to the DOE/HQ for resolution.

The Senior Review Panel provided independent review of the PE analyses. The panel consisted of six highly reputable technical experts from academia, state environmental agencies, and consulting organizations who were familiar with various aspects of radioactive-waste management. Five members of the panel were selected by the DOE; one member was selected by the National Governors Association.

The six-member Review Team included representatives from Sandia National Laboratories and Oak Ridge National Laboratory, DOE contractors, and consulting organizations. This team was responsible for the overall technical quality and consistency of the analysis. The team examined the validity of assumptions, the selection of models and pathways, and the application of the models to the sites being considered.

The Core Teams were composed of technical staff members from Sandia National Laboratories and Oak Ridge National Laboratory. Each Team had six to seven members and was responsible for the analysis of certain sites. This responsibility included information collection, interactions with site technical staff, and analysis. The two teams maintained close communication to ensure consistency between the teams in applying the PE approach.

The site technical staff included staff members who were familiar with details of the environmental characteristics and disposal issues at each of the 15 sites being evaluated. For those sites that had on-going disposal performance assessment activities, site technical staff were those staff members who were extensively involved with the performance-assessment analysis. Site technical staff members were responsible for working with the Core Teams and arranging for interaction with other site personnel. Because interactions with each site resulted in contacting numerous staff, these individuals are not listed in Appendix A but are acknowledged at the end of their respective site chapters in Volume 3 of this report.

The Core Teams worked with representatives from the 15 sites to use important research, site characterization, modeling, and other analyses that had been performed at the sites. These interactions included visits by the Core Teams to each of the sites. As a result of the accompanying information exchange with site personnel, it was possible to incorporate into the PE analyses the best documented understanding of technical staff members who had spent years studying the individual sites.

3.2 QUALITY ASSURANCE/QUALITY CONTROL

The PE quality assurance/quality control (QA/QC) procedures developed by the Core Teams provided traceability of all data, assumptions, and calculations. A fundamental aspect of the QA/QC plan was to document each assumption and calculation. The QA form (shown in Figure 3-2) was used to document every significant piece of data, interaction with site personnel, assumption, reference, and calculation method used in the PE. Each QA form has a unique file name consisting of the site code, topic code, preparer's initials, and date. The QA forms were maintained in easily accessible files throughout the PE process and will be readily available for one year after completion of this study. The forms will then be archived.

SITE NAME:_____ FILE NAME:_____

CONTACT POINT:_____ DATE:_____

_____/_____
(PHONE) (FAX)

DISCUSSION SUMMARY:

REFERENCES:

UPDATE OF PREVIOUS TOPIC: File Name:_____ Date:_____

NAME OF PREPARER:_____

Figure 3-2. Sample QA record.

The quality assurance procedures for the PE ensured that the best documented data were available for the PE and that models were appropriate for site conditions. The Core Teams collected information, worked with site technical staff, and performed the PE analysis while maintaining consistency between the teams in applying the PE approach. The site technical staff worked with the Core Teams and reviewed the site-specific data and assumptions that were used in the PE. The general approach used by the Core Teams was to

1. Collect, both independently and from site technical staff, data and other information including conceptual models used for performance assessments for low-level waste (LLW) disposal.
2. Develop preliminary conceptual models for the disposal site based on assembled data and information.
3. Visit the site for first-hand information and then work on a continuing basis with site technical staff and other technical personnel in developing the PEs.
4. Discuss preliminary conceptual models with the site technical staff and modify as necessary.
5. Define the important pathways and ensure the availability of applicable data.

Through this approach, which was independent of computer models, the Core Teams were able to accommodate in their calculations a wide variety of sites with differing complexities and levels of supporting data.

The Review Team was responsible for the overall quality and consistency of the analysis through examination of the validity of assumptions, the selection of models and pathways, and the application of the models to the sites being considered. As previously indicated, the Senior Review Panel independently reviewed the PE analyses and documented their review.

4. APPROACH AND ASSUMPTIONS USED IN THE PERFORMANCE EVALUATIONS

The goal of the performance evaluation (PE) was to quantify and compare the limitations of 15 DOE sites for the disposal of MLLW. The objective was, therefore, to use a set of modeling assumptions of sufficient detail to capture major site-specific characteristics and yet generic enough for consistent application at all sites. Additionally, the analyses were developed to ensure that no systematic biases were introduced, that the sites were analyzed consistently, and that all major assumptions were clearly stated.

The PE consisted of relatively simple analyses based on the approach used in many low-level waste (LLW) performance assessments. The approach used incorporated a generic framework that was consistent across all sites while, at the same time, it permitted the analysts to accommodate the input of site-specific data and knowledge. When staff members at a site had performed more complex analyses, the relevant components of these analyses were factored into the parameters used in the PE analysis. With this approach, it was possible for the Core Teams to compare sites using a consistent analysis that incorporated the best available data.

The PE teams did not use more complex analyses, such as those found in some performance assessments, for two primary reasons: (1) several of the 15 sites to be evaluated did not have data to support a more sophisticated analysis, and (2) the additional time and resources required to complete such an effort were not consistent with the goals of the PE analysis.

Although the approach was simple, every effort was made to ensure that the PEs were technically adequate for the intended purpose and that the PE results reflected the real strengths and weaknesses of each of the 15 sites for the disposal of MLLW. To ensure technical adequacy, the PE teams adopted the following strategy, that is, they agreed that they would use

- Existing knowledge, analyses, and data at each site to the extent practicable,
- Well-established policies and recommendations on disposal-related issues, and
- Extensive and continuous reviews from both internal and external sources. Internal reviewers were defined as individuals who were familiar with the specific performance evaluation; external reviewers were those who were familiar with radiological assessment, but not the specific evaluation.

The PE was based solely on radiological assessment for disposal even though the wastes under consideration also contain hazardous components that are subject to RCRA requirements (RCRA, 1976). The PE analysis assumed that the chemical components of the wastes would be treated to Land Disposal Restrictions (LDRs) according to RCRA's treatment processes and that a MLLW disposal facility would comply with all RCRA design criteria.

Because the PE was a radiological assessment for disposal, it followed DOE Order 5820.2A (DOE, 1988a) as the basis for the analysis. According to this Order, the DOE is required to dispose of all LLW and MLLW to ensure that it is managed in a manner that assures

protection of the health and safety of the public. The Order requires that a performance assessment be prepared for each proposed DOE disposal facility for LLW and MLLW to demonstrate compliance with certain performance objectives. As specified in the DOE Order, these objectives require the DOE facility operators to

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

In accordance with requirements in the EPA's 40 CFR Part 61, the performance objective for limiting releases of radionuclides to the atmosphere generally is expressed in terms of a limit on dose equivalent of 10 mrem (0.1 mSv) per year. The requirement for protection of groundwater resources, although unspecified in the DOE Order, usually has been interpreted by DOE sites in terms of limits on concentrations of radionuclides that are reasonably consistent with a limit on dose equivalent of 4 mrem (0.04 mSv) per year for the drinking water pathway only, assuming consumption of two liters per day of water. This assumption is consistent with proposed revisions of EPA standards for radioactivity in public drinking water supplies, as specified in 40 CFR Part 141 (EPA, 1991).

Although, as stated above, the performance measures used in the PE were consistent with the performance objectives in DOE Order 5820.2A (DOE, 1988a), their application was slightly different than that used in most *performance assessments.* In a *performance assessment,* the analyst compares the doses estimated from performance calculations for a proposed disposal system (i.e., disposal facility and surrounding environment) with the performance objectives in DOE Order 5820.2A to demonstrate whether the proposed facility is in compliance. However, the purpose of the performance evaluation (PE) was to provide estimates of the maximum concentrations of each radionuclide that would comply with the performance measures. This "waste concentration," attenuated by the natural and man-made barriers in the disposal facility and environment, provided an estimate of the concentration at the performance boundary (i.e., the point of compliance) that was consistent with the performance measure. A maximum waste concentration was associated with each radionuclide and each performance measure. The minimum of these radionuclide-specific and performance-measure-specific waste concentrations was the permissible waste concentration for each radionuclide. (This approach is similar to that used in the Savannah River E-Area performance assessment [MMES et al., 1994].)

The PE teams could have used all of the performance objectives listed in DOE Order 5820.2A (DOE, 1988a) as performance measures. However, as explained more fully in Chapter 5 of this volume, experience with LLW performance assessments has shown that some of the performance objectives are generally more restrictive than others. Specifically, a performance objective of 4 mrem (0.04 mSv) per year for the drinking water pathway (from the groundwater protection requirement) is generally more restrictive than the 25 mrem (0.25 mSv) per year performance objective for all pathways because the major portion of the all-pathways dose is from ingestion of drinking water. Similarly for inadvertent intrusion, the performance objective for chronic exposures of 100 mrem (1 mSv) per year is generally more restrictive than the performance objective for acute exposure (500 mrem [5 mSv]). Thus, the PE performance measures used the more restrictive performance objectives:

- 4 mrem (0.04 mSv) per year from the drinking water pathway for releases to groundwater,
- 10 mrem (0.1 mSv) per year from all exposure pathways for atmospheric releases, and
- 100 mrem (1 mSv) per year from all exposure pathways for long-term, chronic exposure of inadvertent intruders.

The PE teams assumed that the performance measures for the water and atmospheric pathways applied at a performance boundary that was 100 m from the edge of the disposal facility and that all performance measures applied for 10,000 y after disposal. The performance boundary and time period were based on the recommendations of the DOE's Performance Assessment Task Team (Wood et al., 1994a). The DOE formed the Task Team, comprised of experts in performance assessment, to make recommendations on implementing DOE Order 5820.2A (DOE, 1988a) in terms of the disposal of LLW. The Task Team's recommendations were used in the PEs where they were applicable. The performance measures and their implementation in the PE are discussed in more detail in Section 5.1.

The PE teams used two generic disposal facilities: a RCRA-compliant, below-ground trench and a RCRA-compliant, above-ground tumulus. To provide consistency in the evaluation of the 15 sites, it was assumed that the size and shape of each generic facility were the same. More detail is presented in Section 5.2.2.

The PE teams assumed that the waste form was grouted treatment residuals because grout is the most common waste form used for the stabilization of waste. More detail is presented in Section 5.2.1.

The PE teams used the same list of radionuclides for 14 of the 15 sites.* The list was based on the DOE MLLW inventory. Determination of the list is discussed in Section 5.2.3. Using the same list of radionuclides for the sites provided consistency among the site evaluations. An assumption of the PE methodology was that radionuclides were continuously released from the disposal facility. This assumption allowed the effects of dispersion and diffusion in the vadose and saturated zones to be neglected. More detail on this assumption is provided in Section 5.3.1.2.

*As explained in more detail in Chapter 10 of Volume 3, the West Valley Demonstration Project site was analyzed for only 18 radionuclides because these were the only ones expected to be in the MLLW at that site.

The PE teams analyzed releases from a MLLW disposal facility through three pathways—water, atmospheric and inadvertent intrusion. The PE teams represented the attenuation of radionuclides that occurred between the waste in the disposal facility and the performance boundary as a "concentration reduction factor" (CRF). The PE teams used the CRF approach so that intermediate results could be displayed in a transparent fashion that allowed comparisons of the effects of the disposal facility and site on overall performance. This approach also allowed comparisons of results from different sites. The CRFs pertained to transport effects only, assuming non-decaying constituents; the PE teams accounted for the effects of radioactive decay separately in the calculations.

The PE teams used two CRFs to account for the attenuation in the water pathway: a CRF for the source (CRF_{Source}) and a CRF for environmental transport (CRF_{Water}). The CRF_{Source} accounted for the attenuation of the radionuclide concentration in the solid waste and the leachate exiting the disposal facility. It is the ratio of the radionuclide concentration in waste to its resulting concentration in the leachate exiting the disposal facility ($C_{Waste}/C_{Leachate}$). The CRF_{Source} calculation is developed further in Section 5.3.1.1. The CRF_{Water} accounted for attenuation of radionuclide concentration in the leachate exiting the disposal facility and the concentration at the performance boundary. It is the ratio of the radionuclide concentration in the leachate exiting the disposal facility to its concentration in the water at the performance boundary ($C_{Leachate}/C_{Water}$).

The CRF_{Water} was not separated into factors for both the vadose and saturated zones because the PE teams assumed that the vadose zone did not provide a concentration reduction. Because the PE teams used a continuous source model and no dispersion assumption, the concentration reduction factor for the vadose zone was equal to one. The CRF_{Water} calculation is developed further in Section 5.3.1.2.

The PE teams used retarded travel time in the vadose and saturated zones to estimate the amount of radioactive decay occurring at the time of arrival at the performance boundary. This effect is discussed in Section 5.3.1.2.

For the atmospheric pathway, the PE teams used two CRFs to account for the attenuation encompassing diffusion to the ground surface and mixing and dispersion in the atmosphere to the performance boundary. The concentration reduction factor for diffusion in soil (CRF_{Diff}) was conservatively modeled (i.e., using large diffusion constants) in an attempt to bound releases from alternative transport mechanisms (e.g., desiccation cracks, burrowing animals, and root uptake). The CRF_{Diff} is the ratio of the radionuclide concentration in the waste to its resulting concentration at the soil surface (C_{Waste}/C_{SS}). The concentration reduction factor for atmospheric mixing and dispersion (CRF_{Disp}) is the ratio of the concentration at the soil surface to its resulting concentration at the 100-m performance boundary. Gaussian dispersion was the primary concentration-attenuating mechanism used in calculating the CRF_{Disp}. These calculations for the atmospheric pathway are developed further in Section 5.3.2.

The intruder scenarios did not explicitly use a CRF. The PE teams analyzed scenarios for exposure of inadvertent intruders by using factors for converting concentrations of radionuclides

in waste to annual effective dose equivalents. The intruder scenarios are discussed further in Section 5.3.3.

As discussed in Section 5.1, the permissible waste concentrations developed in the PE were based on three separate performance measures related to (1) drinking water, (2) atmospheric releases, and (3) inadvertent intrusion. The PE teams did not estimate the permissible waste concentrations based on a combination of these three performance measures because the simple models used in this analysis did not account for the time history of releases and doses. Rather, the models only provided estimates of the peak releases and doses. Therefore, because the peak doses for the water and atmospheric releases generally occur at different times, adding the two to estimate total doses from all release pathways was not deemed appropriate.

Relative to those used in many performance assessments for LLW disposal facilities, the PE teams used simple and conservative conceptual models for flow and transport of radionuclides. The PE teams used data based on interactions with the technical staff members who were most familiar with characterization and analysis at each site. The data came from a wide range of sources, including site-specific field and laboratory experiments, site-specific analyses, the relevant scientific literature, and site technical staff. The source of each data value used in the analyses is listed in the individual site chapters (see Volume 3 of this report).

Illustrative Example

The PE approach for an exposure pathway involving the consumption of contaminated groundwater is shown in Figure 4-1. The following steps describe the approach used for this exposure pathway:

1. Identify the important radionuclides that are present in the DOE MLLW inventory. This identification is based on those radionuclides common within the DOE complex. Additional details on the radionuclide inventory are presented in Section 5.2.3.

2. For each radionuclide, determine the maximum permissible concentration in the appropriate medium (C_{Water} for water) at the 100-m performance boundary. This step involves the application of the annual effective dose equivalent (represented as the performance measure) and the appropriate pathway dose conversion factors (annual effective dose equivalent per unit concentration) for each radionuclide. Details are provided in Section 5.4.

3. Relate the performance measure to the permissible radionuclide concentration in the leachate ($C_{Leachate}$) released from the disposal facility. Using results of site analyses and data evaluation, estimate the attenuation due to dilution of the radionuclides in the groundwater system from the location below the facility to the performance boundary to calculate the environmental-transport-to-drinking-water CRF (CRF_{Water}) between the two locations. Treat pathways to surface water similarly. Additional details are provided in Section 5.3.1.2.

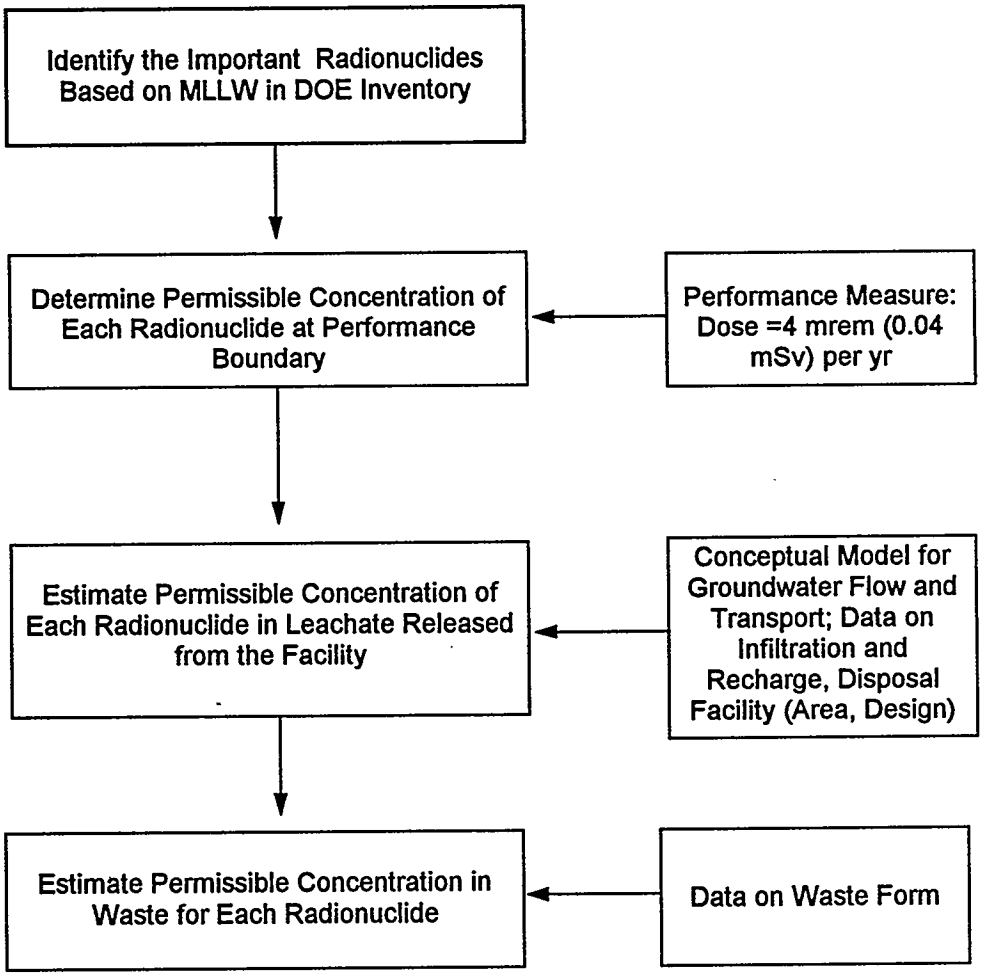


Figure 4-1. Steps in PE methodology for a water pathway.

4. Estimate the attenuation of radionuclide concentrations between the leachate and the original waste form in the disposal facility. A desorption model represents the equilibrium partition of radionuclides between the solid waste form and the liquid phase in contact with the waste form. Additional details are provided in Section 5.3.1.1.

   Determine the radioactive decay term to account for the decrease in activity during the time the waste was detained in the disposal facility plus the delay due to the retarded travel time through the vadose zone and groundwater system to the performance boundary. Incorporation of this term increases the permissible waste concentration because of the decrease in activity occurring as a result of radionuclide decay taking place prior to arrival of the radionuclide at the performance boundary. Additional details are provided in Section 5.4.2.

   Calculate the permissible concentration in the waste for each radionuclide. For the water pathway, the maximum permissible concentration in the waste for a radionuclide is the product of the permissible concentration at the performance boundary, the source-to-leachate CRF (CRF_{Source}), the leachate-to-performance-boundary CRF (CRF_{Water}), and the radioactive decay term. Additional details are provided in Chapter 5.

Details of the PE approach for the water pathway are provided in Section 5.3.1 and Appendix B of this volume.

Atmospheric releases were treated similarly in the PEs; however, a diffusion model was used to estimate the fluxes from the disposal facility to the ground surface, and a Gaussian dispersion model was used to transport the fluxes to the performance boundary. Additional details of the PE approach for atmospheric releases are provided in Section 5.3.2 and Appendix C of this volume.

For intrusion scenarios, the outcomes of the PEs included estimates of permissible radionuclide concentrations using the annual effective dose equivalent (the performance measure) and the scenario-specific dose conversion factors (i.e., annual effective dose equivalents per unit concentration of radionuclides in waste) for appropriate exposure pathways (e.g., inhalation of particulates, ingestion of vegetation, and external exposures). The calculations included a radioactive decay term for the time of intrusion. In making the calculation, assumptions were made about the amount of waste that might be brought to the surface from direct intrusion into the waste. In conducting the PEs, two generic intruder scenarios were used—homesteader and post-closure drilling; since they were controlling, dose estimates were made for chronic exposures only. Additional details of the intruder scenarios are provided in Section 5.3.3.

5. COMPONENTS OF THE PERFORMANCE EVALUATION

Technical processes, assumptions, and equations—either determined by regulations, expert opinions, or scientific information—are used in a performance evaluation. This section presents the technical details of the processes, assumptions, and equations used in the PE and justifications for the assumptions that were used.

The details of the components and assumptions of the PE and their linkage provide a basic understanding of the PE process. Components are defined here as the individual building blocks that describe the phenomena of release, migration, and the resulting exposures of humans to radionuclides from a disposal facility. The basic components of the PE are the following:

- performance measures
- source term
- transport
- exposure pathways and dose conversion factors

5.1 PERFORMANCE MEASURES

In the PE, selected radiological performance measures are used to estimate permissible waste concentrations at a given disposal site. Performance measures, which are analogous to performance objectives for actual disposal systems, as defined in DOE Order 5820.2A (DOE, 1988a), are used with the site-specific PE to estimate the disposal limits for radionuclides. The performance objectives, as defined in the DOE Order, were discussed in Chapter 4:

- 4 mrem (0.04 mSv) per year from the drinking water pathway for releases to groundwater,
- 10 mrem (0.1 mSv) per year from all exposure pathways for atmospheric releases, and
- 100 mrem (1 mSv) per year from all exposure pathways for long-term, chronic exposure of inadvertent intruders.

5.1.1 Implementation of Compliance with the Performance Objectives

Four issues arise in the implementation of compliance with the performance objectives specified in DOE Order 5820.2A (DOE, 1988a). The issues are (1) point of compliance, (2) time period of compliance, (3) protection of groundwater resources, and (4) doses from radon. The following discussions present the rationale used in the PE methodology for resolving these issues.

Point of Compliance. The point of compliance for the performance objectives for off-site individuals and protection of groundwater resources is not specified in DOE Order 5820.2A (DOE, 1988a). In accordance with a recent recommendation of the DOE's Performance Assessment Task Team (Wood et al., 1994a), the boundary between the disposal site and possible locations of off-site individuals is assumed to be defined by a 100-m buffer zone around all waste disposal units at a site. Exposures to off-site individuals then are assumed to occur at the location of maximum concentrations of radionuclides outside the 100-m buffer zone. Similarly, the

performance objective for protection of groundwater resources is assumed to apply at any location outside the 100-m buffer zone (Wood et al., 1994a). Although this assumption would allow radionuclide concentrations in groundwater inside the buffer zone to exceed drinking water standards, performance assessments at DOE low-level waste disposal sites (e.g., see MMES et al., 1994; ORNL, 1994) have indicated that the concentrations inside the buffer zone should not exceed those outside the buffer zone by large amounts. The boundary of the 100-m buffer zone is defined in the PE as the "performance boundary."

Time Period of Compliance. The time period of compliance with the performance objectives is also not specified in DOE Order 5820.2A (DOE, 1988a). Thus, the performance objectives presumably would apply at all future times. However, in accordance with a recommendation of the DOE's Performance Assessment Task Team (Wood et al., 1994a), a time limit of 10,000 y is applied to the performance objectives for off-site individuals, inadvertent intruders, and protection of groundwater resources. This time limit is the same as that specified in EPA standards for disposal of spent fuel, high-level waste, and transuranic waste in 40 CFR Part 191 (EPA, 1992), and it acknowledges that predictions of disposal facility performance for much longer times are highly speculative. A time limit of 10,000 y is expected to ensure an adequate level of protection of public health and the environment and therefore, is used in the PE methodology.

Protection of Groundwater Resources. The performance objectives for LLW disposal do not define what is meant by "protection of groundwater resources," even though this requirement usually is assumed to mean compliance with the limits on radionuclide concentrations as specified in the EPA drinking water standards. In performance assessments for LLW disposal at DOE sites, one of two options for drinking water standards normally is used: (1) concentration limits for radium and gross alpha activity and a limit on dose equivalent to the whole body or any organ of 4 mrem (0.04 mSv) per year for all beta/gamma-emitting radionuclides, as specified in current EPA standards (40 CFR Part 141 [EPA, 1991]), or (2) a limit on effective dose equivalent of 4 mrem (0.04 mSv) per year for all radionuclides. The second option has been adopted in the PE methodology because it is functionally equivalent to the proposed standards for man-made radionuclides (EPA, 1991). Furthermore, the performance objective for groundwater protection usually is assumed to apply only to radionuclides released from the disposal facility, while doses from natural background or other man-made sources not under the control of the DOE disposal site are excluded. This assumption with regard to groundwater protection conforms with longstanding recommendations of the National Council on Radiological Protection and Measurements (NCRP, 1993) and the International Commission on Radiological Protection (ICRP, 1991) that dose limits for members of the public do not include doses from natural background.

Doses from Radon. The performance objectives in DOE Order 5820.2A (DOE, 1988a) do not specify whether doses from radon and its short-lived decay products should be taken into account. However, the performance measures used in the PE (i.e., the dose limits for the water pathway, airborne releases, and exposures of inadvertent intruders) specifically exclude doses from inhalation of radon and its short-lived decay products. This exclusion, which would affect the results of the PE only for isotopes of radium and their uranium and thorium precursors, is based on regulatory precedents for controlling radiation exposures of the public as described below.

In general, exposures of the public to radon are not currently regulated in the same manner as are releases of other radionuclides. For example, current EPA standards for radioactive materials in public drinking water supplies (40 CFR Part 141) (EPA, 1991), for radionuclide releases from operating nuclear fuel-cycle facilities (40 CFR Part 190), and for airborne radionuclide releases from DOE facilities (40 CFR Part 61) (EPA, 1993), all specifically exclude radon. Although both the EPA and the NCRP have recommended indoor radon levels at which remedial actions should be considered, the dose rates accompanying these action levels are far above those for other radiation sources. Proposed revisions of the drinking water standards include a concentration limit for radon (EPA, 1991), but the proposal is highly controversial, due in part to the high cost of complying with the limit, and has not yet been issued as a Final Rule. Finally, in proposed new guidance on radiation protection of the public (EPA, 1994), the EPA excludes doses from radon in the dose limit for members of the public from all sources combined and from any authorized limits for individual practices or sources of exposure. For these reasons, radon is specifically excluded in the PE methodology.

Although 40 CFR 61 excludes radon from the dose limit (EPA, 1993), exposures to radon are controlled by specifying a limit on the release rate to the atmosphere of 20 pCi/m^2-s. Therefore, the assumptions of the PE are that a limit on the release rate of radon could be used as a future design objective for MLLW disposal facilities and that the design objective could be met by an appropriate combination of limits on the concentrations of radon precursors in waste and the thickness of cover materials over the waste.

5.1.2 Basis for Performance Measures Used in the Performance Evaluation

For many LLW disposal sites, releases of radionuclides to off-site locations are assumed to occur primarily via the groundwater or surface water pathway. Acceptable releases then would be controlled by the following two performance objectives: (1) a limit on effective dose equivalent of 25 mrem (0.25 mSv) per year from all exposure pathways, and (2) a limit on effective dose equivalent of 4 mrem (0.04 mSv) per year from the water pathway only. Performance assessments for the DOE LLW disposal sites have indicated that the performance objective for the water pathway only will, in general, be more restrictive than the performance objective for the all-exposure pathway because the terrestrial foodchain pathways are a relatively inefficient way of delivering dose from the use of contaminated water compared with direct ingestion (MMES et al., 1994; ORNL, 1994). Therefore, an effective dose equivalent of 4 mrem per year from the water pathway is used in the PE as the performance measure.

The performance objective for inadvertent intruders specifies two dose limits, one for long-term, chronic exposure scenarios and the other for scenarios involving a single, acute exposure. Performance assessments at the DOE LLW disposal sites have indicated that, for most sites and facility designs, the lower annual dose limit for long-term, chronic exposure scenarios is more restrictive than the higher dose limit for a single, acute exposure in determining acceptable disposal limits (MMES et al., 1994; ORNL, 1994). Therefore, in most cases, the dose limit for acute exposure scenarios can be ignored. The long-term, chronic exposure scenarios for inadvertent intruders used in the PE for MLLW disposal facilities are discussed in Section 5.3.3.1.

5.1.3 Effective Dose Equivalents

The performance measures for evaluating the impacts of radionuclide exposures occurring through the water pathway, the atmospheric pathway, and inadvertent human intrusion are expressed in terms of limits on the effective dose equivalent. The concept of the effective dose equivalent was developed by the ICRP to provide a system that could be used for expressing the risks of partial body exposures (primarily resulting from the deposition of radionuclides in the human body) and whole body exposures (due, for example, to exposures from external sources) on an equal-risk basis. In developing the system, the ICRP (a) based the limits on the risk of death from cancer and hereditary effects; and (b) considered, in the case of internally deposited radionuclides, not only the dose during the year of intake but also the dose resulting from the continuing presence of this material in the body (ICRP, 1979). Conversion of the dose to a portion of the body (as, for example, to a single organ within the body) into an equivalent dose to the whole body is accomplished through use of what are called "tissue weighting factors." The concept of the effective dose equivalent has been adopted by the EPA and conforms with its current radiation protection policies as well as those of other Federal agencies (Office of the President, 1987).

5.1.4 Summary

The following performance measures were selected for the PE analysis in determining acceptable radionuclide concentrations in wastes destined for placement in MLLW disposal facilities:

- For releases of radionuclides to groundwater or surface water beyond the 100-m buffer zone around disposal sites, a limit on effective dose equivalent of 4 mrem (0.04 mSv) per year from the drinking water pathway for 10,000 y after disposal, excluding doses from radon and its short-lived decay products;

- For releases of radionuclides to air beyond the 100-m buffer zone around disposal sites, a limit on effective dose equivalent of 10 mrem (0.1 mSv) per year from all exposure pathways for 10,000 y after disposal, excluding doses from radon and its short-lived decay products; and

- For chronic exposures of inadvertent intruders onto disposal sites following loss of active institutional controls at 100 y after disposal, a limit on effective dose equivalent of 100 mrem (1 mSv) per year for 10,000 y after disposal, excluding doses from radon and its short-lived decay products.

5.2 SOURCE TERM

The source term is defined as the concentrations of the contaminants that are released from the boundary of a disposal facility. Three subcomponents are important in determining the source term: the physical and chemical characteristics of the waste form, the characteristics of the disposal facility, and the characteristics of the radionuclides. Each of the three subcomponents used in the PE is described below.

5.2.1 Waste Form

As discussed in Section 1.1, the DOE's MLLW will be treated according to RCRA's Land Disposal Restriction treatment requirements prior to disposal. MLLW with organic hazardous components will likely undergo some type of thermal treatment followed by stabilization of the residues. For those MLLW containing metals, immobilization is the most common treatment process. The Site Treatment Plans developed for each DOE site will specify the preferred type of treatment for each mixed waste stream. The final waste forms in many cases are not specified in the STPs; therefore, assumptions are made in the PE about the waste form.

The waste form assumed in the PE is grouted treatment residuals. Grout, consisting primarily of hydrated Portland cement and fly ash, is often used to stabilize wastes containing hazardous metals or to stabilize residues resulting from thermal treatment. Grout is considered the primary waste-form stabilizer in three LLW performance assessments: Oak Ridge SWSA 6 (ORNL, 1994); Hanford vaults (Kincaid et al., 1993); and Savannah River Z-Area (MMES et al., 1992). Although other waste forms may be proposed in the STPs, grouted treatment residuals will probably be the most common. Additional information and assumptions used to evaluate the behavior of a grouted waste form are presented in Section 5.3.1.1.

5.2.2 Disposal-Facility Design

Disposal-facility design assumptions are divided into three categories: type and size of facilities, engineered barriers, and location of facilities. Six DOE sites currently have plans to develop MLLW facilities (DOE, 1994); as described in the following sections, the design features proposed for these facilities have been taken into consideration in determining the assumptions used in developing the PE.

5.2.2.1 Facility Types and Sizes

Two generic disposal facilities are considered in the PE: a RCRA-compliant, below-ground trench and a RCRA-compliant, above-ground tumulus. Using these generic facilities allows common assumptions for consistent comparison of the disposal capabilities of the 15 sites. Assumptions pertaining to these facilities are presented in this section. For sites that are currently developing plans for an MLLW disposal facility and have advanced their plans to the point of specifying a facility design that is significantly different from the generic designs used in the PE, a third site-specific design (see Section 5.2.2.2) is also evaluated.

Compliance with RCRA is assumed by designing the disposal facility in the manner prescribed in the RCRA regulations. The major RCRA requirements on disposal facility design are the specification of a cover system with low hydraulic conductivity and the requirement for a leachate collection system with a liner. The liner and leachate collection system are assumed to function as designed for 30 y, the minimum period of active monitoring required (RCRA, 1976). The cover system is assumed to function for 100 y, the period of active institutional controls defined in DOE Order 5820.2A (DOE, 1988a). These assumptions are discussed more fully in Section 5.2.2.2.

Generic Trench

A schematic representation of the RCRA-compliant, below-ground trench is shown in Figure 5-1. The generic trench is assumed to be a square, 2500-m^2 facility based on designs assumed in the draft Environmental Management Programmatic Environmental Impact Statement. This facility is small enough to be located on all 15 sites currently being analyzed in the PE. Grouted waste is assumed to be placed directly into the trench and then backfilled with natural soils. The mixing fraction (i.e., the volume of waste contained in a unit volume of disposal facility) is assumed to be 2/3 and accounts for the non-waste volume between trenches and from packaging of the grout blocks. This fraction relates the radionuclide concentration in the leachate to that in disposed waste (see Section 5.3.1.1).

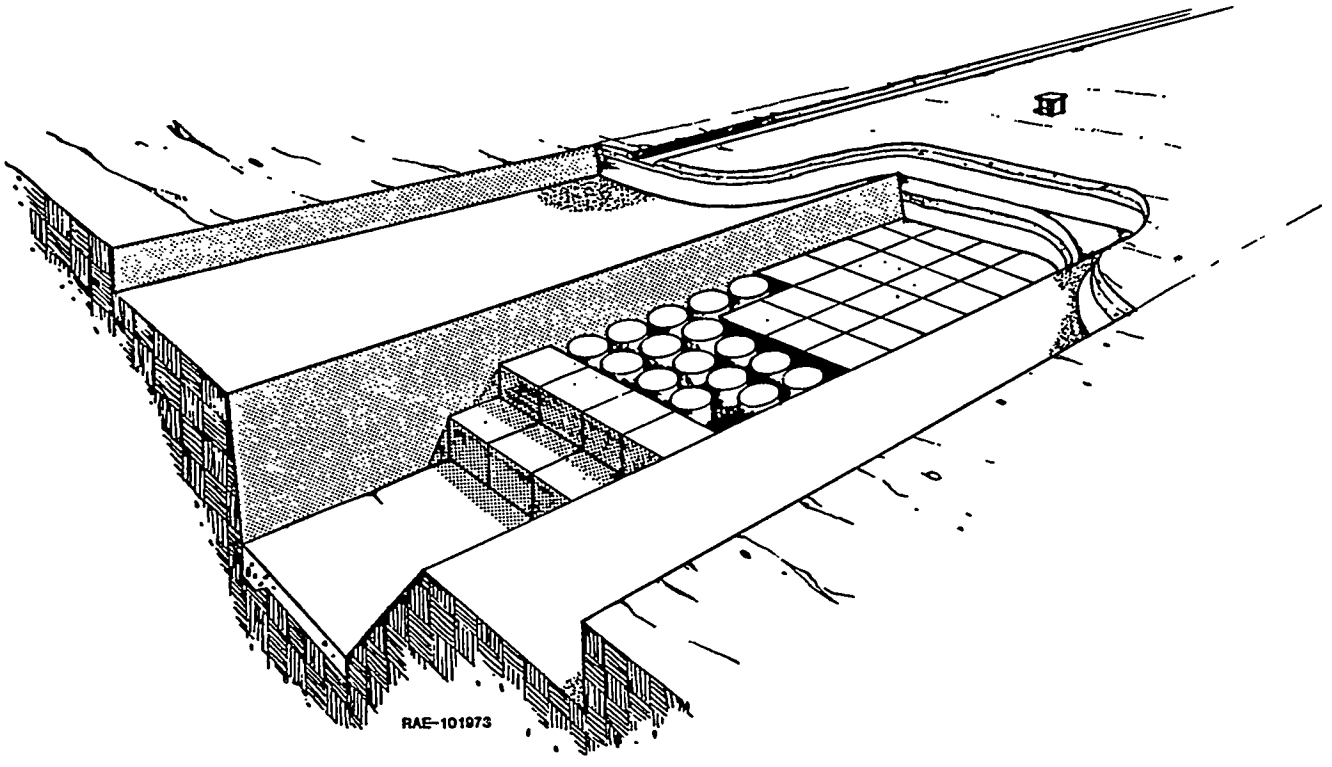


Figure 5-1. Artist's conception of a generic RCRA-compliant, below-ground trench.

Generic Tumulus

A schematic representation of the RCRA-compliant, above-ground tumulus is shown in Figure 5-2. Like the generic trench, the generic tumulus is also assumed to be a square, 2500-m^2 facility. Grouted waste is assumed to be placed within the concrete vault boxes and sealed. The vault boxes are then stacked three high and covered with a RCRA-compliant cover system. The mixing fraction is assumed to be 1/3. This value is based on the volumetric fraction of waste contained in a concrete vault used in the tumulus design in the Oak Ridge SWSA 6 performance assessment (ORNL, 1994).

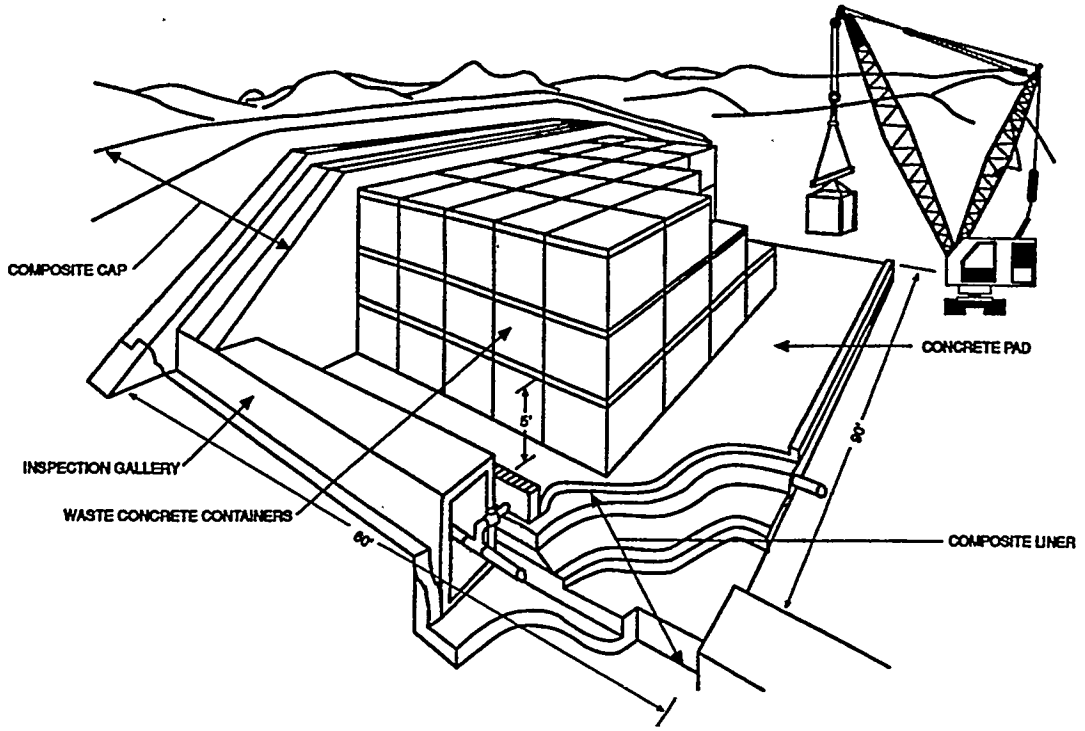


Figure 5-2. Artist's conception of a generic RCRA-compliant, above-ground tumulus.

5.2.2.2 Engineered Barriers

Engineered barriers are man-made features emplaced in a disposal facility to enhance its waste isolation capabilities. In the PE analysis, engineered barriers apply only to the water pathway. The atmospheric release analysis assumes that no credit is taken for the differences in disposal facility features (see Section 5.3.2), and the inadvertent intruder analyses use different assumptions for the engineered barriers (see Section 5.3.3).

Three components regarding the performance of the engineered barriers are considered in the PE: liner and leachate collection system, cover system, and concrete vaults for the tumulus design. The behavior of the major engineered components of the generic facilities is assumed to be the same across all 15 sites, regardless of site-specific climate and other mechanisms promoting or retarding degradation. Figure 5-3 shows the assumed time-dependent performance of the engineered barriers for the generic trench and tumulus facilities.

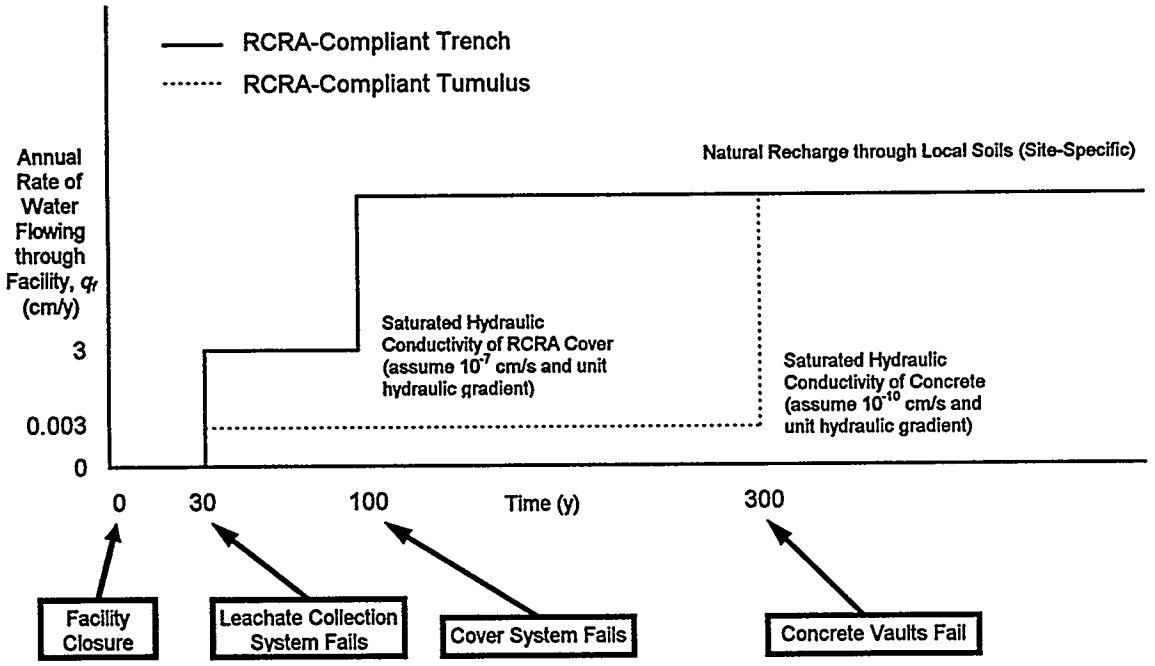


Figure 5-3. Assumed performance of engineered barriers for the two generic facilities.

Generic Trench

For the generic trench, a low-permeability RCRA cover and a RCRA leachate collection system are assumed operable at the time of closure. For the first 30 y following closure, the RCRA leachate collection system is assumed to collect all leachate from the facility so that no releases from the site occur during this time (Figure 5-3).

The liner and leachate collection system are assumed to fail abruptly at 30 y after closure. At that time, releases of radionuclides from the facility are possible through infiltration of water through the RCRA cover into the facility. The rate of water moving into the facility is calculated based on the lesser of either a unit hydraulic gradient and a saturated hydraulic conductivity of 1×10^{-7} cm/s (as required by RCRA for the cover system), or the site-specific value for natural recharge through local soils.

Institutional controls are assumed to last for 100 y (see Section 5.1). Beyond this time, when the site caretaker no longer maintains the facility cover, the rate of water moving through the facility is assumed to be the same as the average annual recharge through local soils. In other words, after 100 y the RCRA cover system and the liner are assumed to have degraded in such a way that they are indistinguishable from the local soils.

Generic Tumulus

For the generic tumulus, a low-permeability RCRA cover system and a RCRA leachate collection system are assumed operable at the time of closure (Figure 5-3). For the first 30 y following closure, the RCRA leachate collection system is assumed to collect all leachate from the facility so that no releases from the site occur during this time.

As with the generic trench, the liner and leachate collection system of the generic tumulus are assumed to fail abruptly at 30 y. At that time, the rate of water moving through the facility is assumed to be based on the lesser of the hydraulic conductivity of concrete vaults at a unit hydraulic gradient or natural recharge through local soils. The saturated hydraulic conductivity of concrete is assumed to be 1×10^{-10} cm/s (0.003 cm/y), which is much less than the hydraulic conductivity of a RCRA cover (3 cm/y). The hydraulic integrity of the concrete vaults in the tumulus facility is assumed to be maintained for 300 y.

Three hundred years after closure, the concrete vaults are assumed to fail hydraulically. The resulting rate of water moving through the facility and through the contacted waste is assumed to be the same as the natural recharge through local soils.

Site-Specific Design

In addition to the two generic designs, for sites with current plans for MLLW disposal facilities, a third, site-specific analysis was also performed using the PE methodology. However, most existing MLLW plans were either trench or tumulus designs; therefore, only sites with a facility design having an assumed performance of engineered barriers that was considerably different than the generic trench or tumulus were evaluated. The results of these analyses are contained in Appendix B of Volume 3 of this report.

5.2.2.3 Location of Disposal Facility

The location assumed for the evaluation of the generic and site-specific facilities is based on site-specific input acquired during site visits by the Core Teams. For sites with MLLW disposal plans, the planned location of the facility is used for the PE analysis. For sites with no existing LLW or planned MLLW disposal facilities, considerations such as current operations, hydrogeology, and future land use were used in designating a proposed location for use in the PE analysis. Because the performance boundary for the analysis is only 100 m from the disposal facility, some larger sites with relatively uniform hydrogeologic conditions selected an approximate location with an appropriate hydrologic setting for the disposal facility.

5.2.3 Radionuclides Evaluated in the PE

In developing the list of radioactive materials used for the PE, specific radionuclides expected to be in MLLW were identified and cross-checked. The initial list was based on data received from the sites represented in the DOE's Mixed Waste Inventory Report (MWIR). These data provided a master list of all radionuclides reported in MLLW in the DOE complex. The master list of radionuclides was screened and checked in the following manner:

1. Radionuclides with half-lives less than five years were eliminated. One hundred years of institutional controls ensure little impact to intruders from such radionuclides. In a similar manner, engineered barriers and travel times ensure little impact to off-site individuals and resources..

2. The screened list was compared to the list of radionuclides considered in the analyses for the EM PEIS, which in turn was based on information in the 1991 Integrated Data Base (DOE, 1992; communication with Bruce Wilkens, Argonne National Laboratory-East, August 26, 1994). The screened list was found to include all radionuclides considered in the EM PEIS with half-lives greater than five years (Table 5-1).

3. The screened list was compared to the list of radionuclides considered in performance assessments for LLW at the Oak Ridge Reservation (ORNL, 1994), the Savannah River Site (E Area vaults) (MMES et al., 1994), the Hanford Site (grout vaults) (Kincaid et al., 1993), the Idaho National Engineering Laboratory (Maheras et al., 1994), and the Nevada Test Site (Area 5) (Magnuson et al., 1992). Included in the performance assessments for these 5 sites were an additional 12 radionuclides with half-lives greater than 5 y that were not reported in the MWIR. These radionuclides were added to the screened list (Table 5-1).

4. Kr-85, a noble gas listed only in the MWIR, was not expected to be present in waste that has been treated for disposal and was therefore eliminated in the final screening.

The final list of radionuclides considered in the PE consists of 58 radionuclides (Table 5-1). No attempt was made to associate parent radionuclides and radioactive decay products; if radioactive decay products were listed explicitly in any of the sources of information, they were included in the master list. Because the effects of any radioactive decay products of significance were considered in developing the dose conversion factors for the various pathways, the effects of any such radionuclides are automatically taken into consideration in the PE calculations (see Section 5.4.2).

Table 5-1. Radionuclide Inventory for Performance Evaluations (screened for half-lives of 5 y or greater) (Part 1 of 2)

	Nuclide	Half-Life (y)[1]	Source of Information						
			MWIR	EM PEIS	PA				
					ORR[2]	SRS[3]	HAN[4]	NTS[5]	INEL[6]
1	H-3	1.23 E+01	*	*	*	*	*		*
2	C-14	5.73 E+03	*		*	*	*		*
3	Al-26	7.30 E+05			*	*			
4	Si-32	1.00 E+02	*						
5	Cl-36	3.01 E+05	*		*				
6	K-40	1.28 E+09	*						*
7	Co-60	5.27 E+00	*	*	*	*	*		
8	Ni-59	7.60 E+04	*	*		*			*
9	Ni-63	1.00 E+02	*	*	*	*	*		
10	Se-79	6.50 E+04	*			*	*		
11	Sr-90	2.91 E+01	*	*	*	*	*	*	*
12	Zr-93	1.50 E+06	*			*			
13	Nb-93m	1.61 E+01	*			*			
14	Nb-94	2.00 E+04	*	*			*		
15	Tc-99	2.13 E+05	*		*	*	*		*
16	Pd-107	6.50 E+06				*			
17	Ag-108m	1.30 E+02	*						
18	Cd-113m	1.41 E+01	*		*	*			
19	Sn-121m	5.50 E+01	*			*			
20	Sn-126	1.00 E+05	*			*	*		
21	I-129	1.57 E+07	*			*	*		*
22	Cs-135	2.30 E+06	*			*	*		
23	Cs-137	3.02 E+01	*	*	*	*	*	*	
24	Ba-133	1.05 E+01	*						
25	Sm-151	9.00 E+01	*	*		*			
26	Eu-152	1.35 E+01	*		*		*		
27	Eu-154	8.59 E+00	*	*	*	*	*		
28	Pb-210	2.23 E+01	*			*			
29	Ra-226	1.60 E+03	*		*	*	*		*
30	Ra-228	5.76 E+00	*	*					
31	Th-229	7.30 E+03	*		*	*			

[1] GENE, 1989
[2] ORNL, 1994
[3] MMES et al., 1994
[4] Kincaid et al., eds., 1993
[5] Magnuson et al., 1992
[6] Maheras et al., 1994

Table 5-1. Radionuclide Inventory for Performance Evaluations (screened for half-lives of 5 y or greater) (Part 2 of 2)

	Nuclide	Half-Life (y)[1]	Source of Information						
			MWIR	EM PEIS	PA				
					ORR[2]	SRS[3]	HAN[4]	NTS[5]	INEL[6]
32	Th-230	7.54 E+04	*		*	*	*		*
33	Th-232	1.40 E+10	*	*	*	*			*
34	Pa-231	3.28 E+04				*			
35	U-232	7.00 E+01	*		*	*			
36	U-233	1.59 E+05	*		*	*			
37	U-234	2.46 E+05	*		*	*	*		*
38	U-235	7.04 E+08	*	*	*	*	*	*	*
39	U-236	2.34 E+07	*		*	*			
40	U-238	4.47 E+09	*	*	*	*	*	*	*
41	Np-237	2.14 E+06	*		*	*	*		*
42	Pu-238	8.77 E+01	*	*	*	*	*	*	
43	Pu-239	2.41 E+04	*	*	*	*	*	*	*
44	Pu-240	6.56 E+03	*	*	*	*	*	*	
45	Pu-241	1.44 E+01	*	*		*	*		
46	Pu-242	3.75 E+05	*		*	*	*		
47	Pu-244	8.00 E+07				*			
48	Am-241	4.33 E+02	*	*	*	*	*		
49	Am-243	7.37 E+03	*		*	*			
50	Cm-243	2.91 E+01			*	*	*		
51	Cm-244	1.81 E+01	*	*	*	*	*		
52	Cm-245	8.50 E+03				*			
53	Cm-246	4.76 E+03				*			
54	Cm-247	1.56 E+07				*			
55	Cm-248	3.48 E+05				*			
56	Cf-249	3.51 E+02			*	*			
57	Cf-250	1.31 E+01				*			
58	Cf-251	9.00 E+02				*			

[1] GENE, 1989
[2] ORNL, 1994
[3] MMES it al., 1994
[4] Kincaid et al., eds., 1993
[5] Magnuson et al., 1992
[6] Maheras et al., 1994

5.3 TRANSPORT

The transport component of the PE addresses the migration and attenuation of radionuclides from the time they are released from the disposal facility to the ultimate location of compliance (performance boundary). Radionuclides are transported from the disposal facility to human receptors through three pathways—water, atmospheric, and inadvertent intrusion.

The PE provides estimates of permissible radionuclide concentrations, C_W, in disposed MLLW treatment residuals at 15 sites consistent with the performance objectives of DOE Order 5820.2A (DOE, 1988a). As discussed in Chapter 4, the radionuclide concentrations in the waste are calculated by the following factors: performance measures, dose conversion factors, concentration reduction factors, and a radioactive decay term.

The calculation for the water pathway is

$$C_{W\text{-}Water} = 1000 \times H_{Water} \times \frac{1}{PDCF_{Water}} \times CRF_{Source} \times CRF_{Water} \times r_{Decay} \quad (5\text{-}1)$$

where

$C_{W\text{-}Water}$ is the concentration of a specific radionuclide in the disposed waste corresponding to the permissible drinking water dose at the performance boundary ($\mu Ci/m^3$);

1000 is a factor for converting $\mu Ci/L$ to $\mu Ci/m^3$;

H_{Water} is the performance measure of 0.004 rem (0.04 mSv) per year effective dose equivalent from consumption of drinking water;

$PDCF_{Water}$ is the dose conversion factor for the water ingestion pathway represented by the annual effective dose equivalent per unit concentration for drinking water for each radionuclide ([rem/y]/ [μCi/L]);

CRF_{Source} is the concentration reduction factor for the source ([mCi/L/[mCi/L]);

CRF_{Water} is the concentration reduction factor for environmental transport in groundwater or surface water ([mCi/L]/[mCi/L]); and

r_{Decay} is the term accounting for radioactive decay during detention time in the waste plus retarded travel time via environmental transport in the water pathway to the exposure point at the performance boundary (dimensionless).

The calculation for the atmospheric pathway is

$$C_{W-Atm} = H_{Atm} \times \frac{1}{SDCF_{Atm}} \times CRF_{Diff} \times CRF_{Disp} \times r_{Decay} \quad (5\text{-}2)$$

where

$C_{W\text{-}Atm}$ is the concentration of a specific radionuclide in the disposed waste corresponding to the permissible dose at the performance boundary due to atmospheric releases ($\mu Ci/m^3$);

H_{Atm} is the performance measure of 0.01 rem (0.1 mSv) per year for all pathways for atmospheric releases;

$SDCF_{Atm}$ is the scenario dose conversion factor represented as the annual effective dose equivalent per unit concentration for all pathways from atmospheric releases for each radionuclide ([rem/y]/[μCi/m^3]);

CRF_{Diff} is the concentration reduction that occurs from the top of the disposal facility to the soil surface as the radionuclide diffuses upward [(μCi/m^3)/ (μCi/m^3)];

CRF_{Disp} is the concentration reduction that occurs as the radionuclide is emitted to the atmosphere, mixed with the ambient air, and dispersed downwind to the performance boundary [(μCi/m^3)/ (μCi/m^3)]; and

r_{Decay} is the term accounting for radioactive decay during detention time in the waste plus travel time via environmental transport in the atmospheric pathway to the performance boundary (dimensionless).

The calculation for the inadvertent intruder pathway is

$$C_{W-Intr} = H_{Intr} \times \frac{1}{SDCF_{Intr}} \times r_{Decay} \tag{5-3}$$

where

C_{W-Intr} is the concentration of a specific radionuclide in the disposed waste corresponding to the permissible dose to an inadvertent intruder (μCi/m^3);

H_{Intr} is the performance measure of 0.1 rem (1 mSv) per year effective dose equivalent for inadvertent intrusion;

$SDCF_{Intr}$ is the annual effective dose equivalent per unit concentration for an intrusion scenario for each specific nuclide ([rem/y]/ [μCi/m^3]); and

r_{Decay} is the decay term accounting for radioactive decay prior to the assumed intrusion (dimensionless).

The PE results based on these general equations estimate the waste concentration in direct correspondence to the permissible dose for each radionuclide, assuming it contributes the entire dose.

5.3.1 Water Pathway

Two CRFs are calculated for the water pathway: one for the attenuation between the waste and leachate exiting the bottom of the disposal facility, CRF_{Source} ; and one for the attenuation between the leachate exiting the disposal facility and the water at the performance boundary, CRF_{Water}. As shown in Equation 5-1, radioactive decay is accounted for by r_{Decay}, and the ingrowth of radiologically significant decay products is accounted for by $PCDF_{Water}$. Therefore, the concentration reduction factors, CRF_{Source} and CRF_{Water}, represent concentration reductions for the radionuclides, not taking radioactive decay into account.

5.3.1.1 Source CRF

For a stabilized waste form (see Section 5.2.1), the source CRF, CRF_{Source}, is defined as

$$CRF_{Source} = C_{Waste} / C_{Leachate} \tag{5-4}$$

where

C_{Waste} is the concentration in the grouted waste form for each radionuclide averaged over the entire volume of waste in the disposal facility (μCi/L), and

$C_{Leachate}$ is the corresponding concentration in the leachate for each radionuclide as it exits the bottom of the disposal facility (μCi/L).

The partitioning of radionuclides between the solid phase (i.e., radionuclides sorbed onto the grout) and the liquid phase (i.e., radionuclides dissolved in the pore water) is assumed to be determined by the equilibrium sorption phenomenon. This assumption is consistent with analyses in the three LLW performance assessments that have evaluated grouted waste forms (ORNL, 1994; MMES et al., 1994; Kincaid et al., 1993). With this assumption, the radionuclide concentration in the leachate based on desorption in infiltrating water can be described (ORNL, 1994) by

$$C_{Leachate} = \frac{C_{Waste}\, f_m}{(\theta_G + K_d^G \rho_G)} \tag{5-5}$$

where

θ_G is the volumetric water content of the grouted waste form (mL/mL);

K_d^G is the distribution coefficient (i.e., solid/liquid partition coefficient) of the radionuclide in the grout (mL/g);

ρ_G is the dry bulk density of the grouted waste form (g/cm^3); and

f_m is the mixing fraction, defined as the ratio of the volume of waste disposed in a unit volume of the facility.

Combining Equations 5-4 and 5-5 derives a relationship for CRF_{Source} in terms of the grout distribution coefficient (K_d^G) and the properties of the stabilized waste (θ_G, ρ_G, and f_m):

$$CRF_{Source} = \frac{\left(\theta_G + K_d^G \rho_G\right)}{f_m} \tag{5-6}$$

In addition to K_d^G, the parameters that affect the source term CRF_{Source} include the dry bulk density of the grout (ρ_G), the volumetric water content of the grout (θ_G), and the mixing fraction (f_m). These terms are summarized in Table 5-2 and are used in all site analyses.

Table 5-2. Summary of Grout and Facility Parameters That Affect the Source Term CRF (CRF_{Source})

Parameter	Value	Comment
Grout Dry Bulk Density (ρ_G)	1.76 g/cm^3	The dry bulk density of grout is defined as the oven-dried mass per unit volume of grout. Value is based on Oak Ridge SWSA 6 performance assessment (ORNL, 1994).
Grout Volumetric Water Content (θ_G)	0.3	The volumetric water content of the grout is defined as the volume of water per unit volume of grout. The grout is assumed to be saturated, which reflects the hygroscopic nature of cementitious grouts. Under saturated conditions the water content and the porosity are the same.
Mixing Fraction (f_m)	Trench 2/3 Tumulus 1/3	This factor is the fraction of waste volume in the disposal facility. The f_m for a trench design is 2/3, and the f_m for a tumulus design is 1/3 (see Section 5.2.2.1).

Limited information exists on appropriate values for radionuclide distribution coefficients in grout (K_d^G); this is due in part to the wide variety of methods used to measure waste leachability from different waste forms. For example, the EPA (1989) identified nine extraction procedures and three different leaching tests. In light of this variability, conservative K_d^G values (i.e., low K_d^G values that yield relatively high radionuclide concentration estimates in the leachate) are used in the PE. These values are based largely on the analysis in the Oak Ridge SWSA 6 performance assessment (ORNL, 1994). The grouted waste form considered in the formulation of K_d^G values in the Oak Ridge performance assessment was based on mixing dry waste with pumpable grout. Conversely, K_d^G values in the Hanford vault performance assessment (Kincaid et al., 1993), and the Savannah River Z-Area vaults performance assessment (MMES et al., 1992), are based on grout formulations in which the radionuclides are contained in water mixed into the grout material. Much higher K_d^G values were used in the performance assessments for the Savannah River and Hanford sites than for Oak Ridge. The lower values used in the Oak Ridge performance assessment are more conservative in that they result in smaller values for CRF_{Source}; thus, these values are used as the primary basis for estimating the values used in the PE. Because of the high degree of variability associated with K_d^G, all values used in the PE are rounded to the nearest order of magnitude (e.g., a K_d^G value of 7 mL/g was assumed to be 10 while a K_d^G of 3000 mL/g was assumed to be 1000) so as not to imply more precision in these values than is justified. The distribution coefficient values used in the PE and the resulting CRF_{Source} for the generic tumulus and trench facilities are listed in Table 5-3.

Table 5-3. Waste K_d^G Values Used in the Performance Evaluation and Resulting Source Concentration Reduction Factors (CRF_{Source}) for the Tumulus and Trench Design (Part 1 of 2)

Radionuclide	Grout K_d (mL/g)	CRF_{Source}	
		Tumulus	Trench
H-3	0	0.90	0.45
C-14	10	54	27
Al-26	100	540	270
Si-32	100	540	270
Cl-36	0	0.90	0.45
K-40	1	6.2	3.1
Co-60	100	540	270
Ni-59	10	54	27
Ni-63	10	54	27
Se-79	1	6.2	3.1
Sr-90	100	540	270
Zr-93	10	54	27
Nb-93m	10	54	27
Nb-94	10	54	27
Tc-99	1	6.2	3.1
Pd-107	100	540	270
Ag-108m	100	540	270
Cd-113m	100	540	270
Sn-121m	10	54	27
Sn-126	10	54	27
I-129	1	6.2	3.1
Cs-135	10	54	27
Cs-137	10	54	27
Ba-133	10	54	27
Sm-151	10	54	27
Eu-152	10	54	27
Eu-154	10	54	27
Pb-210	100	540	270
Ra-226	10	54	27
Ra-228	10	54	27
Th-229	100	540	270
Th-230	100	540	270
Th-232	100	540	270
Pa-231	100	540	270

Table 5-3. Waste K_d^G Values Used in the Performance Evaluation and Resulting Source Concentration Reduction Factors (CRF_{Source}) for the Tumulus and Trench Design (Part 2 of 2)

Radionuclide	Grout K_d (mL/g)	CRF_{Source}	
		Tumulus	Trench
U-232	100	540	270
U-233	100	540	270
U-234	100	540	270
U-235	100	540	270
U-236	100	540	270
U-238	100	540	270
Np-237	100	540	270
Pu-238	100	540	270
Pu-239	100	540	270
Pu-240	100	540	270
Pu-241	100	540	270
Pu-242	100	540	270
Pu-244	100	540	270
Am-241	100	540	270
Am-243	100	540	270
Cm-243	100	540	270
Cm-244	100	540	270
Cm-245	100	540	270
Cm-246	100	540	270
Cm-247	100	540	270
Cm-248	100	540	270
Cf-249	100	540	270
Cf-250	100	540	270
Cf-251	100	540	270

A review of Table 5-3 reveals two interesting results pertaining to the values for CRF_{Source} for the trench and tumulus. First, the tumulus values are twice the trench values. This result is due solely to the differences in mixing fraction values selected for the two facilities. Second, the values for H-3 and Cl-36 are less than one, implying a concentration increase for these radionuclides. This effect is the result of an assumed K_d^G value of zero for these two radionuclides. This low K_d^G value implies that these radionuclides are not sorbed on the grouted waste form but rather they are entrained in the matrix. These radionuclides will tend to readily leave the solid phase and enter the water phase very easily.

5.3.1.2 Environmental Transport CRF for Water

The environmental transport CRF for water, CRF_{Water}, describes the attenuation of radionuclide concentrations between the leachate and the performance boundary (100 m) in the groundwater or surface water. The environmental transport CRF for water is defined as

$$CRF_{Water} = C_{Leachate} / C_{Water} \quad (5\text{-}7)$$

where

CRF_{Water} is the environmental transport concentration reduction factor for the water pathway (dimensionless);

$C_{Leachate}$ is the radionuclide concentration in the leachate exiting the disposal facility (μCi/L); and

C_{Water} is the resulting radionuclide concentration in water at the performance boundary (μCi/L).

To conduct the water pathway analyses, a generic conceptual model has been developed that incorporates site-specific geometry and water flow paths into a simple transport analysis (Figure 5-4). This generic conceptual model is applicable to both arid and humid sites. Different hydrogeologic environments cause the water flow to vary considerably from one site to another; therefore, site-specific development of conceptual models for water flow relies heavily on interaction with the site technical staff in determining the pathways to be included in the PE. An acceptable conceptual model and its associated assumptions must be supported by site-specific knowledge and data. Depending on site-specific conditions, surface water, the vadose zone, and groundwater may be pathways of consideration. Once the water flow paths are established, the same radionuclide transport assumptions are imposed on each site to provide consistency in the PE analyses. Several specific details about the generic conceptual model are described in Appendix B.

Major Assumptions for Flow and Transport in the Generic Conceptual Model

A number of assumptions were made with respect to flow and transport in the generic model. The more important of these are

- Steady-state flow in the vadose and saturated zones (see Appendix B of this volume).
- Continuous and constant source release from the disposal facility with step increases at the times of failure of the engineered barriers (see Section 5.2.2.2).
- One-dimensional flow and transport in the vadose zone with no lateral spreading from diffusion or dispersion (see Appendix B of this volume). As a result of this assumption, the CRF for transport through the vadose zone is unity.
- When the relevant geologic formations in the vadose zone at a site were known to contain fractures, these fractured sections were not considered in the analysis and the resulting thickness of the vadose zone was reduced.
- One-dimensional flow and transport in the saturated zone with no diffusion or hydrodynamic dispersion in the longitudinal or transverse directions (see Appendix B).

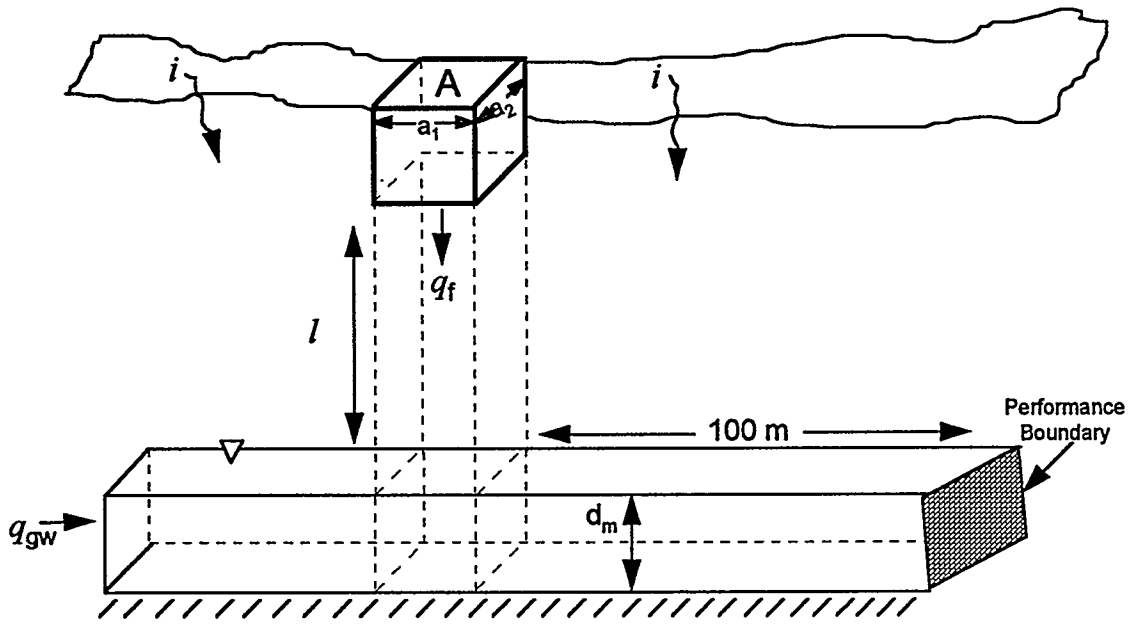


Figure 5-4. Generic conceptual model for the water pathway.

- When the relevant geologic formations in the saturated zone were known to contain fractures, matrix porosity and the fractured porosity were combined as an effective porosity.
- Complete mixing across an appropriate mixing depth used to represent dilution in the saturated zone. The appropriate mixing depth is based on aquifer properties; thin aquifers are assumed to be completely mixed across their entire depth. A one-dimensional flow, three-dimensional transport code (PAGAN) (Chu et al., 1991) is used to estimate an appropriate mixing depth for thicker aquifers, using estimates for dispersivities compiled by the Electric Power Research Institute (EPRI, 1985) based on the type of geologic formation in the saturated zone.
- Sorption of dissolved radionuclides on the porous media in both unsaturated and saturated zones result in retarded travel times of radionuclides.
- Solubility constraints are not considered.

The most significant PE assumption that simplifies the estimation of the CRF_{Water} is the continuous release of radionuclides from the disposal facility. This assumption is justified for two reasons. First, the radionuclide inventory in the disposal facility is unknown, so the duration of release is unknown; assuming a continuous release source is therefore conservative relative to assuming a release of finite duration. Second, the assumption allows the effects of longitudinal dispersion and diffusion on peak concentration in the vadose and saturated zones to be neglected. With a non-decayed, continuous release source, the concentration at the base of the vadose zone eventually equals the leachate concentration, and the concentration in groundwater at the performance boundary eventually equals the concentration within the leachate-groundwater mixing zone. Mechanical dispersion in the longitudinal direction affects the shape of the

contaminant front but provides no attenuation of the peak. Mechanical dispersion in the lateral transverse direction is a much weaker (10 to 100 times less) effect than dispersion in the longitudinal direction except under low-flow conditions (Freeze and Cherry, 1979, p. 396). While lateral transverse dispersion provides some attenuation of the centerline concentration peak, the effect is minor. Therefore, ignoring the attenuating effect of lateral transverse dispersion is a simplifying assumption that provides conservatism. Additional discussion on this subject is provided in Appendix B of this volume.

Vadose Zone

The volumetric flow of water through the facility that generates leachate is based on the performance and size of the disposal facility as discussed in Sections 5.2.2.1 and 5.2.2.2. It is defined as

$$Q_f = q_f A \tag{5-8}$$

where

Q_f is the volumetric flow rate of leachate from the disposal facility (m^3/y);
q_f is the rate of water moving through the facility (m/y); and
A is the facility plan area (m^2).

As shown in Figure 5-3, q_f is initially controlled by the RCRA cover (trench) or the concrete vault (tumulus) properties. When all engineered barriers have failed, q_f is assumed to be equal to the natural recharge through local soils, i (Figure 5-4).

No lateral spreading is assumed, so the leachate flux through the vadose zone is confined to the soil column directly below the plan area of the facility. Additional details about the effects of this assumption are provided in Appendix B of this volume. The water pathway analysis considered the effects of fracture flow where relevant (see Section 6.1.2.4 in this volume).

The water travel time in the vadose zone, t_{wv}, for steady-state, one-dimensional flow under unit gradient conditions is defined as

$$t_{wv} = \frac{l\theta_W}{q_f} \tag{5-9}$$

where

l is the distance between the disposal facility and groundwater (m), and
θ_W is the volumetric moisture content in the vadose zone (mL/cm^3).

Retarded contaminant travel time in the vadose zone, t_{cv}, is defined as

$$t_{cv} = t_{wv}\,(1 + K_d\,\rho_b / \theta_W) \tag{5-10}$$

where

K_d is the soil distribution coefficient for the radionuclide in the vadose zone (mL/g), and
ρ_b is the dry bulk density of the media in the vadose zone (g/cm^3).

Saturated Zone

As contaminated water enters the aquifer, the contaminant mixes with clean groundwater, forming a plume with a shape controlled by aquifer and contaminant properties. The generic conceptual model assumes that instantaneous and complete mixing within the aquifer occurs directly below the facility.

The dilution of contaminants by mixing with groundwater that was originally free of any radionuclides is the only attenuation effect in the water pathway, and the CRF is (see Appendix B)

$$CRF_{Water} = (Q_f + Q_{gw}) / Q_f \tag{5-11}$$

where Q_{gw} is the volumetric groundwater flow (m^3/y), defined as

$$Q_{gw} = q_{gw}\, d_m\, a_2 \tag{5-12}$$

where

q_{gw} is the groundwater Darcy velocity, the volume discharge per unit bulk area (m/y),
d_m is the contaminant mixing depth in the groundwater (m), and
a_2 is the width of the facility (m).

The contaminant mixing depth in the groundwater, d_m, is estimated according to the properties of the aquifer. For thin aquifers, mixing over the entire aquifer thickness is assumed. For thicker aquifers, the mixing depth is estimated by the plume thickness at 100 m from the facility due to vertical dispersion. The estimate is provided through use of a one-dimensional flow, three-dimensional transport computer code (PAGAN) [Chu, et al., 1991].

The water travel time in the saturated zone, t_{ws}, assuming steady-state, one-dimensional flow, is defined as

$$t_{ws} = \frac{(100)n}{q_{gw}} \tag{5-13}$$

where n is the porosity of the saturated zone.

Retarded contaminant travel time in the saturated zone, t_{cs}, is defined as

$$t_{cs} = t_{ws}\,(1 + K_d\,\rho_b / n). \tag{5-14}$$

The radionuclide-specific arrival time equals the detention time in the disposal facility plus the retarded travel times in the vadose and saturated zones.

Radioactive Decay Term

For release to water, the radioactive decay term, r_{Decay}, accounts for radioactive decay during detention time in the disposal facility plus retarded travel to the performance boundary. The radioactive decay term is defined as

$$r_{Decay} = exp\left[\frac{ln(2)(t_a)}{t_{1/2}}\right] \quad (5\text{-}15)$$

where

t_a is the radionuclide arrival time at the performance boundary, calculated as the detention time in the disposal facility plus retarded travel time (y); and

$t_{1/2}$ is the radionuclide half-life (y).

Site-Specific Data Requirements

The minimum site-specific data required to perform the water pathway analysis are listed in Table 5-4.

Table 5-4. Site-Specific Data Required for Water Pathway Analyses

Parameter	Description
i	Natural recharge through local soils (m/y)
q_{gw}	Darcy flow rates in saturated zone (m/y)
n	Porosity in saturated zone (cm^3/cm^3)
θ_w	Ambient moisture content in the unsaturated zone (mL/cm^3)*
d_m	Mixing depth in the aquifer (m)
ρ_b	Dry bulk density of the porous media in the unsaturated and saturated zones (g/cm^3)
K_d	Solid/liquid partition coefficient of the porous media in the unsaturated and saturated zones (mL/g)

* Because one milliliter of water is essentially one cubic centimeter, this unit is dimensionless.

5.3.2 Atmospheric Pathway

Radionuclides can be transported from the waste disposal unit to the soil surface through a variety of mechanisms including diffusion, burrowing animals, plant root uptake, and water flow through cracks caused by desiccation. When radionuclides reach the soil surface, they may be entrained in the air in vapor form (volatiles) or suspended in the air by the wind as particulate matter (non-volatiles). Once airborne, the radionuclides are assumed to be transported via atmospheric dispersion to a receptor located 100 m downwind of the disposal facility.

Because of the difficulty and uncertainty associated with modeling the variety of potential transport processes for radionuclides to the soil surface, the approach taken in the PE is to use volatile or gaseous diffusion with several conservative assumptions to bound the releases from all transport processes. The diffusion mechanism has been used in two existing low-level waste performance assessments to analyze the atmospheric pathway (Kincaid et al., 1993; MMES et al., 1994), and the burrowing animal mechanism has been used by one existing LLW performance assessment (Maheras et al., 1994). The plant root uptake and desiccation crack mechanisms have not been considered in existing performance assessments.

Only the volatile radionuclides, H-3 and C-14, are considered in the PE for atmospheric releases. The radionuclide, I-129, may become volatile during the high temperatures of a reactor accident but is not generally volatile under disposal-facility conditions.

Two environmental transport CRFs are used for the atmospheric pathway. The concentration reduction that occurs from the top of the disposal facility to the soil surface as the radionuclide diffuses upward, CRF_{Diff}, is defined as:

$$CRF_{Diff} = \frac{C_{Waste}}{C_{SS}} \tag{5-16}$$

where

C_{Waste} is the radionuclide concentration in the waste ($\mu Ci/m^3$), and
C_{SS} is the radionuclide concentration at the soil surface above the disposal facility resulting from the concentration in the waste ($\mu Ci/m^3$).

The concentration reduction that occurs as the radionuclide is released into the air and dispersed downwind to the performance boundary, CRF_{Disp}, is defined as:

$$CRF_{Disp} = \frac{C_{SS}}{C_{Atm}} \tag{5-17}$$

where

C_{Atm} is the radionuclide concentration in the ambient air resulting from the air concentration at the soil surface ($\mu Ci/m^3$).

As was shown in Equation 5-2, radioactive decay, r_{Decay}, is separately accounted for. Therefore, the concentration reduction factors for atmospheric releases represent concentration reductions for the radionuclides, not taking radioactive decay into account.

5.3.2.1 Transport Calculations

The analytical method to estimate the two atmospheric concentration reduction factors is summarized below, discussed in detail in Appendix C of this volume, and illustrated in Figure 5-5. To bound the amount of a particular radionuclide transported to the soil surface by various transport processes with diffusion, certain assumptions are incorporated into the analyses. These assumptions are the following:

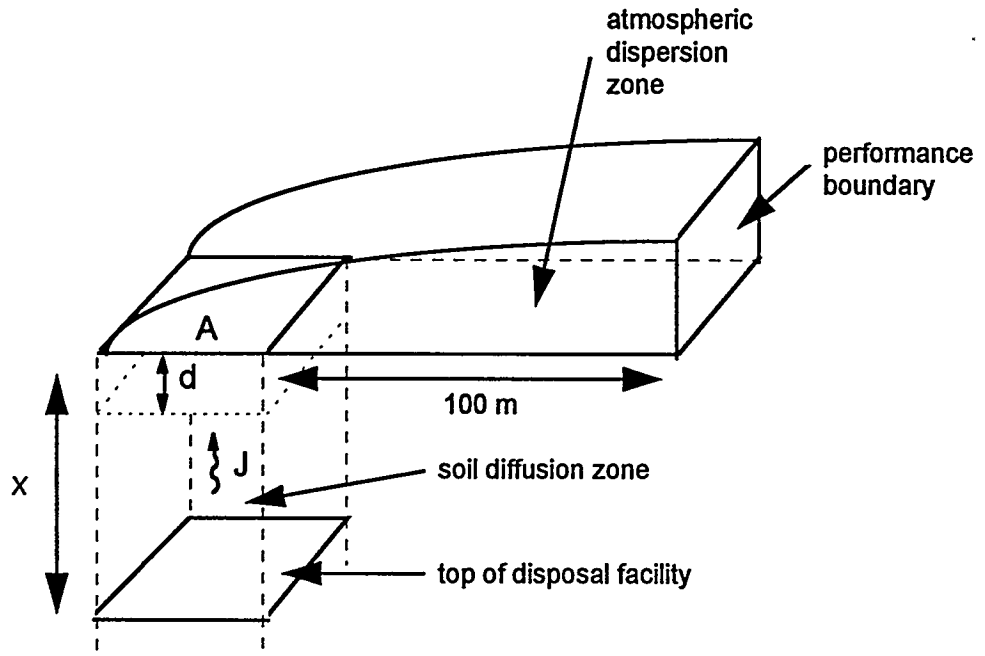


Figure 5-5. Conceptual model for the atmospheric pathway.

- The waste is completely mixed within the soil, and therefore, no credit is taken for the waste disposal facility.
- An upper-bound diffusion coefficient is used in the calculation.
- All diffusion is upward.
- Diffusive flux is approximated as first order, linear, and steady state.

The CRF_{Diff} is calculated using the following equation:

$$CRF_{Diff} = \frac{xns}{rd} \tag{5-18}$$

where

x is the cover thickness above the disposal facility (m);
n is the porosity of the soil (dimensionless);
s is the percent saturation of the soil (dimensionless);
r is the ratio of the density of water in air to that in the liquid phase, for ^{3}H, or the ratio of the CO_2 concentration in the air to that in dissolved water, for ^{14}C (dimensionless); and
d is the depth of the surface soil (m).

The calculation of CRF_{Disp} is

$$CRF_{Disp} = \frac{d}{DA_D} \tag{5-19}$$

where

D is the ^3H or $14CO_2$ diffusion coefficient in air (m^2/s); and
A_D is the atmospheric dispersion term ([$\mu Ci/m^3$] / [$\mu Ci/m^2$-s]).

The atmospheric dispersion term, A_D, is estimated using a Gaussian air dispersion model. Site-specific meteorological data are used in the analysis.

To estimate transport time of the radionuclide to the soil surface, the diffusion velocity, v_d, is approximated as a one-dimensional, first-order, linear expression. Assuming v_d is uniform, the transport time, t_d, is given by

$$t_d = \frac{x^2}{D}. \tag{5-20}$$

Based on the parameter values presented in the next section, the transport time through the soil is estimated to be in the range of a few days. Transport time in the atmosphere is in the range of seconds to minutes. Therefore, the assumed detention time in the disposal facility of 100 y is clearly the dominating factor in determining the time before arrival of any releases at the performance boundary.

For releases to the atmosphere, r_{Decay} accounts for radioactive decay during detention in the disposal facility, and travel to the soil surface and the performance boundary. The radioactive decay term is the same as that defined in Equation 5-15 for the water pathway, where

t_a is the radionuclide arrival time at the performance boundary, calculated as the detention time in the disposal facility plus travel time (y).

5.3.2.2 Data Requirements

The key parameters that affect the atmospheric pathway include the cover thickness (x), the depth of surface soil (d), the soil porosity (n), the percent saturation in the soil voidspace (s), the diffusion coefficient in air (D), and the atmospheric dispersion parameter (A_D). These terms are summarized in Table 5-5 and are used in all site analyses.

Table 5-5. Data Required for Atmospheric Pathway Analyses (parameters in bold type represent generic values)

Parameter	Comments
Cover thickness (*x*)	**For the generic trench and tumulus, the soil thickness is equal to 1 m, based on the SRS E-Area performance assessment (MMES et al., 1994). For site-specific facilities, a site-specific cover thickness is used.**
Depth of surface soil (*d*)	**The depth of the surface soil, assumed to be 0.01 m. This value was chosen based on the INEL RWMC PA (Maheras et al., 1994).**
Soil porosity (*n*)	The fraction of the soil above the waste disposal facility that is void space.
Percent saturation in the soil void space (*s*)	The fraction moisture content in the void space of the soil above the waste disposal facility.
Diffusion coefficient in air (*D*)	**This parameter is used in the volatile radionuclide calculations to determine flux density. For ^{3}H, *D* is equal to 2.39×10^{-5} m^2/s, and for $^{14}CO_2$, *D* is equal to 1.40×10^{-5} m^2/s.**
Atmospheric dispersion parameter (A_D)	The atmospheric dispersion parameter is the radionuclide concentration of a particular radionuclide at 100 m beyond the facility boundary per unit flux density. This value is obtained by dispersion models and depends on site-specific meteorological parameters (i.e., wind speed and direction, ambient air temperature, and atmospheric stability).

5.3.3 Inadvertent Intrusion

As previously discussed, the performance measure in the PE against which to gauge the intruder dose is the 100 mrem (1 mSv) per year from long-term, chronic exposure. Because future societal behavior is difficult to predict, the scenarios for inadvertent intrusion used in this analysis are based on the assumption that future inadvertent intruders behave similarly to such individuals at the present time.

5.3.3.1 Scenario and Pathway Models

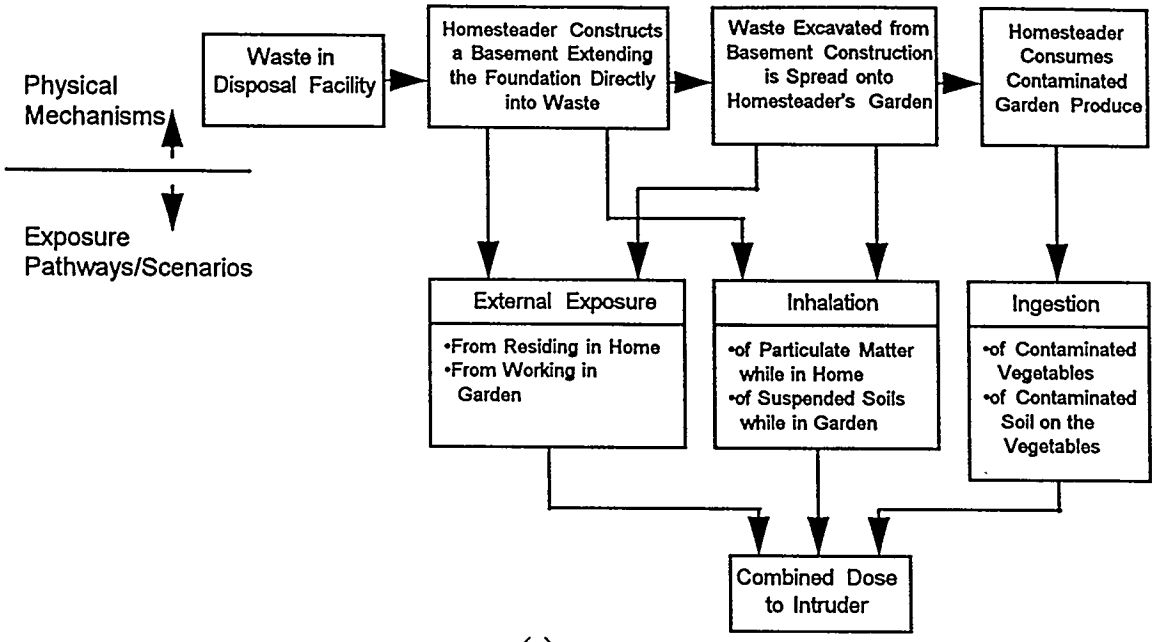
Based primarily on assumptions used by the NRC in developing the waste classification system for near-surface disposal of radioactive waste in 10 CFR Part 61 (NRC, 1982), several standard scenarios for inadvertent intrusion involving an initial acute exposure followed by a long-term chronic exposure have been widely used. These scenarios include a basement construction (acute) and homesteader or agriculture (chronic) scenario; a discovery in which an inadvertent intruder immediately recognizes the presence of the disposal facility, promptly closes the excavation, and withdraws (acute) and a resident, non-homesteader (chronic) scenario in which the engineered barriers are assumed to remain intact and prevent direct access to waste; and a drilling (acute) and post-drilling (chronic) scenario.

Performance assessments at DOE LLW disposal sites (MMES et al., 1994; ORNL, 1994) have revealed the following insights about exposure scenarios for inadvertent intruders. First, scenarios for long-term, chronic exposure usually are more important than those for acute

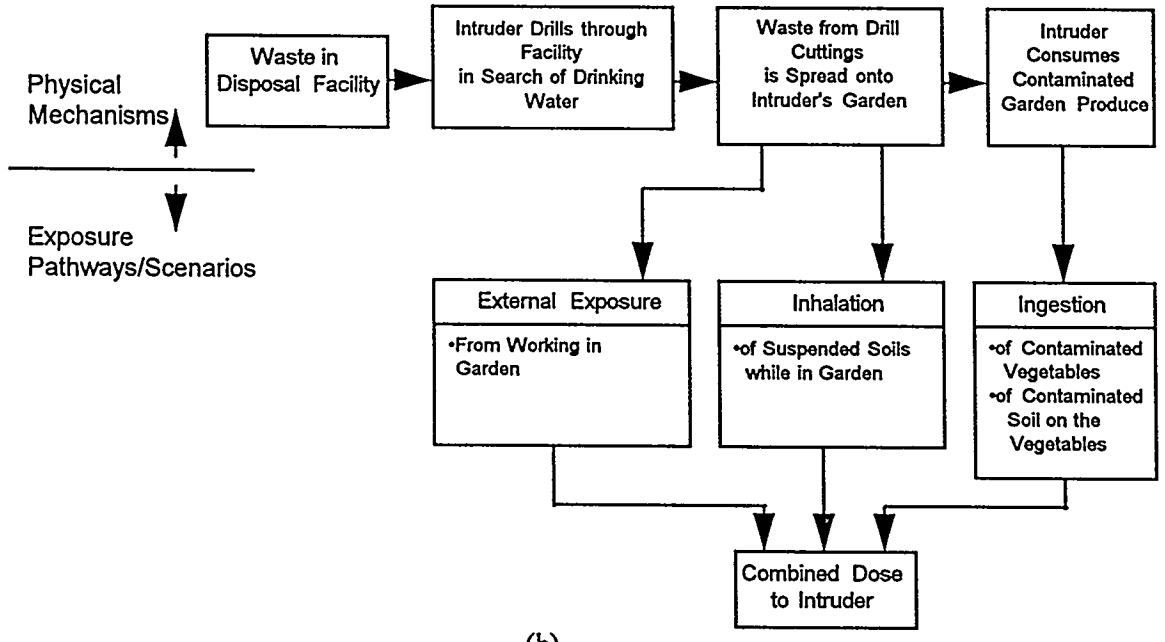
exposure in determining limits on concentrations of radionuclides that would be acceptable for disposal in order to meet the performance objective for protection of inadvertent intruders. This effect is primarily due to the short duration of exposure and the higher permissible dose limit for acute exposures. Second, the chronic homesteader scenario usually determines the concentration limits for most radionuclides, although the post-drilling scenario can be more restrictive for relatively short-lived radionuclides (e.g., Sr-90 and Cs-137) if drilling into waste could reasonably occur well before a large scale excavation into the waste. This effect is due to the presence of engineered barriers that would preclude excavation but not drilling. Consequently, the PE analysis is based on the homesteader and post-drilling scenarios; they are described below. Implicit in this discussion is the understanding that the actual long-term living habits of future inadvertent intruders are difficult to predict with any certainty and that this uncertainty presumably increases with time. The obvious difficulty of predicting the behavior of future societies has been recognized by others (e.g., NAS, 1995). The exposure scenarios for inadvertent intruders discussed below are consistent with the approach used by the NRC in developing the waste classification system in 10 CFR Part 61 and are expected to bound the exposures from common hypothesized inadvertent intrusions.

Homesteader Scenario. After active institutional control ceases, the homesteader scenario is normally based on the assumption that an intruder comes onto the disposal site and establishes a permanent homestead, including on-site sources of water and foodstuffs. Waste in the disposal units is assumed to be accessed when the intruder constructs a home directly on top of the disposal facility and the foundation of the home extends into the waste. Some of the waste exhumed from the disposal facility during excavation for the foundation then is assumed to be mixed with native soil in the intruder's vegetable garden. Once a permanent homestead has been presumed to have been established on the site, various pathways for chronic exposure are assumed, including ingestion of vegetables grown in the contaminated garden soil; direct ingestion of contaminated soil; external exposure to contaminated soil while working in the garden or residing in the home on top of the disposal facility; and inhalation, while working in the garden or residing in the home, of radionuclides released into air from contaminated soil. The various exposure pathways assumed in the homesteader scenario are illustrated in Figure 5-6(a).

Post-Drilling Scenario. The post-drilling scenario assumes that an intruder who resides permanently near the site drills through a disposal unit in constructing a well for a domestic water supply. It is further assumed that, following construction of the well, the contaminated material brought to the surface during drilling operations is mixed with native soil in the intruder's vegetable garden. In this case, the relevant pathways for chronic exposure include ingestion of vegetables grown in the contaminated garden soil, ingestion of contaminated soil, and external and internal exposures, the latter occurring primarily through inhalation, while working in the garden. These pathways essentially are the same as those assumed for the homesteader scenario, except that the concentrations of radionuclides in garden soil resulting from intrusion by drilling are generally assumed to be significantly less (e.g., by about an order of magnitude) than the concentrations in garden soil resulting from a large-scale excavation at the site as in the homesteader scenario. The various exposure pathways assumed in the post-drilling scenario are illustrated in Figure 5-6(b).



(a)



(b)

Figure 5-6. Exposure pathways in the PE for (a) homesteader and (b) post-drilling intrusion scenarios.

The exposure pathways assumed in the post-drilling scenario often are the same as some of the pathways assumed in the homesteader scenario. However, as indicated previously, the amount of waste assumed to be mixed with contaminated soil in the intruder's vegetable garden usually is assumed to be significantly less in the post-drilling scenario. Furthermore, certain exposure pathways involving residence in a home on top of the disposal facility are assumed in the homesteader scenario but are not relevant in the post-drilling scenario. Therefore, if the two scenarios are assumed to occur at the same time after disposal (i.e., if a large-scale excavation and drilling into the waste are assumed to become credible at the same time), the homesteader scenario always will result in higher estimates of dose per unit concentration of radionuclides in the disposed waste and it would thus impose more restrictive limits on concentrations of radionuclides that would be acceptable for disposal. However, if the post-drilling scenario occurred before the homesteader scenario (e.g., if, for some period of time after loss of active institutional controls, the engineered barriers precluded a large-scale excavation into the waste but not drilling), the post-drilling scenario could result in higher estimates of dose per unit concentration for relatively short-lived radionuclides (e.g., Sr-90 and Cs-137). These radionuclides would decay appreciably between the times assumed for first occurrence of the post-drilling and homesteader scenarios.

An important characteristic of the standard scenarios for inadvertent intrusion that were developed by the NRC in 10 CFR Part 61 (NRC, 1982) and that have been used in performance assessments of DOE LLW disposal sites is that the scenarios are reasonably generic (i.e., they are generally applicable to many disposal sites and facility designs). Thus, estimates of dose to inadvertent intruders are generally less dependent on site-specific conditions than are estimates of dose from off-site releases into groundwater.

The definitions of exposure scenarios and assumptions about exposure pathways and model parameter values used in the performance assessments to date have considered, to a large degree, the design of the disposal facility (e.g., the expected long-term integrity of engineered barriers that might preclude intrusion into the waste, placement of waste at depths sufficient to preclude intrusion by surface excavation) and local environmental conditions affecting reasonable actions that might be taken by inadvertent intruders (although future social behavior, including intrusion scenarios, is resistant to prediction). Because the generic disposal facilities in this analysis are the same for all sites, the intrusion-based permissible waste concentrations are also generally the same for all sites. The exception is for site environmental conditions that dictate different times for intrusion.

Inadvertent intrusion is assumed to be precluded during any period of active institutional controls over a disposal site. In accordance with the NRC's 10 CFR Part 61 (NRC, 1982) and DOE Order 5820.2A (DOE, 1988a), an assumption of the PE is that active institutional controls over MLLW disposal facilities will be maintained for 100 y after disposal. Thus, for any facility design that does not include engineered barriers or other features (e.g., disposal well below the ground surface) to deter intrusion into the waste, inadvertent intrusion could occur at any time after loss of active institutional controls (i.e., any time beyond 100 y after disposal).

The time of intrusion depends on the method of intrusion, the disposal technology, and site-specific factors. Unless modified by site-specific conditions, intrusion in the homesteader scenario is assumed to occur at 500 y for the generic above-ground tumulus and at 300 y for the generic shallow trench. The waste form is assumed to be indistinguishable from its surroundings at these times. Additional credit is given to the tumulus because of the concrete vault-boxes incorporated into this design. Unless modified by site-specific conditions, the post-drilling scenario is assumed to occur after the 100 y of institutional control when the waste is assumed to be indistinguishable from natural formations. The assumed lifetime of engineered barriers for preventing intrusion is consistent with that used by the NRC in developing the waste classification system in 10 CFR Part 61 (NRC, 1982).

An assumption of standard exposure scenarios for inadvertent intrusion is that an intruder receives exposures from use of contaminated water from an on-site well, in addition to exposures from direct intrusion into solid waste within the disposal facility. In the PE, however, only scenarios involving direct intrusion into solid waste in the disposal facility are considered. This simplification to the analysis is based on performance assessment results (MMES et al., 1994; ORNL, 1994) that indicated that the maximum exposures to radionuclides obtained from an on-site well generally would not occur at the same time as the maximum exposures to radionuclides in solid waste in the disposal facility. Thus, the total dose to an inadvertent intruder would not be the sum of the maximum doses for the two pathways. In some cases, the total dose would be well approximated by the greater of the doses from either pathway.

Furthermore, the dose limit in the performance measure for the water pathway at the boundary of the 100-m buffer zone (4 mrem/y) is considerably more stringent than the dose limit in the performance measure for protection of inadvertent intruders (100 mrem/y). Therefore, at most sites, the maximum dose from use of contaminated water obtained from an on-site well should be only a small fraction of the dose limit for inadvertent intruders. Finally, focusing only on direct intrusion into solid waste conforms with the original intent of the concept of a hypothetical inadvertent intruder at waste disposal sites. This concept was intended to be used in determining limits on residual concentrations of radionuclides in disposal facilities (NRC, 1982), by considering exposures of off-site individuals as the basis for determining limits on permissible releases of radionuclides from disposal facilities. Thus, the approach used in the PE provides a clear separation between a performance measure for the water pathway that is used to limit permissible releases from disposal facilities and a performance measure for inadvertent intruders that is used to limit permissible concentrations in the facilities even if no releases occurred.

In principle, evaluations of exposures of future inadvertent intruders to solid waste in disposal facilities should take into account depletions of radionuclide inventories in the waste over time because of release and transport from the disposal facility as well as radioactive decay. However, because models for estimating radionuclide releases often are intended to be conservative (i.e., to overestimate releases that might occur), the resulting inventories in the disposal facility and, thus, the resulting doses to inadvertent intruders, could be underestimated. Therefore, in intruder dose analyses, the conservative assumption often is made that radionuclide inventories in solid waste are depleted only by radioactive decay; this assumption is used in the PE.

5.3.3.2 Assumptions and Generic Approach Used in the PE

In summary, the assumptions related to the intruder scenarios are as follows:

- Only chronic exposure is considered (associated with a 100 mrem [1 mSv] per year performance measure).
- Intrusion occurs only beyond the period of active institutional controls (100 y), and depends on the intrusion scenario and site-specific factors. Benefits from passive institutional controls (e.g., permanent marker system, public records of prior land use) after active institutional control are not considered.
- Only pathways related to direct intrusion into the waste are considered (i.e., water pathways are not considered).
- Inhalation exposure to radon is not considered (see Section 5.1.1).
- The performance measures apply for 10,000 y (see Section 5.1). Therefore, radionuclides with significant long-lived decay products and whose doses increase with time (i.e., U-233, U-234, U-235, U-238, Th-230, Pu-244, Cm-245, and Cm-247) are taken at the maximum dose within the 10,000 y.
- Depletion of radionuclide inventory from the waste disposal facility by leaching through water pathways is not considered.

The generic equation for the permissible concentration in the waste (C_w) based on an intrusion scenario for a specific radionuclide is as follows:

$$C_{W-Intr} = H_{Intr(x)} \times \sum_i \frac{1}{SDCF_{Intr,i}} \times r_{Decay} \qquad (5\text{-}21)$$

where

$C_{W\text{-}Intr}$ is the concentration of radionuclide in the waste disposal unit ($\mu Ci/m^3$);
$H_{Intr(x)}$ is the total intruder annual dose for scenario x (rem/y);
$\sum_i SDCF_{Intr,i}$ is the radionuclide-specific scenario dose conversion factor [(rem/y)/($\mu Ci/m^3$)]; and
r_{Decay} is the term accounting for radioactive decay prior to intrusion (dimensionless).

The total intruder doses consist of doses from several exposure pathways (e.g., direct external exposure to waste, ingestion of food or soil, and inhalation of suspended airborne soil particles). Factors that are used to calculate the doses, and therefore the scenario dose conversion factors (*SDCF*s), include, for example, correction factors for the amount of natural soil mixed with exhumed waste, plant-to-soil concentration ratios, shielding factors, and the fraction of time the individual is exposed through a certain pathway. Some of the factors used to calculate the *SDCF* are constant, while others may vary depending on factors such as exposure pathway and disposal technology. A more detailed discussion on the intruder exposure pathways and associated factors is provided in Appendix D of this volume.

To calculate the $C_{W\text{-}Intr}$ in Equation 5-21, the annual dose $H_{Intr(x)}$ is set equal to the performance measure (i.e., 0.1 rem [1 mSv] per year chronic dose). The total *SDCF* for an intruder scenario is the sum of each *SDCF* for each exposure pathway applicable to an intruder scenario, as shown in Equation 5-21. Further details are presented in Appendix D of this volume. Each *SDCF* is an annual effective dose equivalent per unit concentration of waste and is in units of (rem/y)/(μCi/m^3). The *SDCF*s for an assumed exposure scenario for inadvertent intruders are derived from existing performance assessments.

The radioactive decay term for the inadvertent intrusion scenarios accounts for radioactive decay prior to an assumed intrusion. Similar to Equation 5-15 for the water and atmospheric pathways, the radioactive decay term is defined as

$$r_{Decay} = exp\left[\frac{ln(2)(t_i)}{t_{1/2}}\right] \tag{5-22}$$

where

t_i is the time of intrusion for the intruder scenario being analyzed (y) (discussed in Section 5.3.3.1); and

$t_{1/2}$ is the radionuclide half-life (y).

The exceptions for the generic intruder scenarios are eight radionuclides with significant long-lived decay products and whose doses increase over time. Six of the radionuclides (U-233, U-234, U-235, U-238, Pu-244, and Cm-247) continue to increase in dose beyond the 10,000-y performance period. Therefore, t_i is assumed to be at 10,000 y when the dose would be at its maximize during that period. Two other radionuclides (Th-230 and Cm-245) also increase in dose over time but have maximum doses before the 10,000-y performance period. The maximum doses for Th-230 and Cm-245 occur at 9,000 and 1,000 y, respectively, which are used as the t_i values.

5.4 EXPOSURE PATHWAYS AND DOSE CONVERSION FACTORS

In estimating the dose from the ingestion or inhalation of radionuclides, factors for converting radionuclide intakes into committed effective dose equivalents are needed. Similarly, in estimating the dose from external exposure, factors for converting concentrations of radionuclides in environmental media (i.e., air, water, and soil) to effective dose-equivalent rates are needed. These factors usually are referred to as dose conversion factors.

5.4.1 Dose Conversion Factors

In the PE, dose conversion factors developed by the EPA (Eckerman et al., 1988; Eckerman and Ryman, 1993) for internal and external exposure of a reference adult are used. The dose conversion factors for internal exposure are based on dosimetric and metabolic models recommended in ICRP Publications 30 and 48 (ICRP, 1979; ICRP, 1986). The dose conversion factors for external exposure include a calculation for exposure to contaminated surface soil, which is a potentially important exposure pathway for inadvertent intruders.

Dose conversion factors for internal and external exposure also have been developed by the DOE (1988b, 1988c). These dose conversion factors are similar to those developed by the EPA. Some differences in the values for internal exposure are due primarily to the approach used in rounding off the estimates; the values for external exposure in the DOE compilation were developed by a calculational methodology that results in slight overestimates of dose. In addition, the DOE compilation of external dose conversion factors does not include values for exposure to contaminated surface soil. Therefore, the DOE compilations are not used in the PE.

In estimating disposal limits for radionuclides based on the performance measure for protection of groundwater and surface water resources, the limits on radionuclide concentrations in water at the performance boundary beyond the 100-m buffer zone are obtained from the dose limit in the performance objective (i.e., a limit on effective dose equivalent of 4 mrem [0.04 mSv] per year), an assumed intake of water of 2 L per day, and the ingestion dose conversion factors developed by the EPA (Eckerman et al., 1988).

In the PE, the concentration limits for radionuclides in water are calculated by using the ingestion dose conversion factors developed by the EPA (Eckerman et al., 1988) rather than by obtaining these limits directly from proposed revisions of the drinking water standards (EPA, 1991). This approach is preferred primarily because the results are reproducible. In addition, the proposed drinking water standards do not include similar calculations for alpha-emitting radionuclides (for which separate concentration limits were proposed [EPA, 1991]), but the same dose limit from the drinking water standards for beta/gamma-emitting radionuclides was selected for the PE. Thus, in the PE, a consistent method for estimating concentration limits in water is used for all radionuclides. For alpha-emitting radionuclides, the assumed dose limit of 4 mrem (0.04 mSv) per year is considerably more restrictive than the concentration limits proposed by the EPA (1991), which are 20 pCi/L (0.7 Bq/L) for Ra-226 and Ra-228; 20 mg/L for uranium based on considerations of chemical toxicity; and 15 pCi/L (0.6 Bq/L) for gross alpha activity, excluding Ra-226, uranium, and Rn-222.

For ingestion of contaminated water, the following general equation shows the factors used to convert the performance measure into a permissible concentration in the drinking water (Eckerman et al., 1988):

$$C_{Water} = H_{Water} \Big/ \left(IR_{Water} * DCF_{Ing} \right) \qquad (5\text{-}23)$$

where

C_{Water} is the radionuclide-specific permissible concentration in drinking water (μCi/L);
H_{Water} is the annual dose (effective dose equivalent) (rem/y);
IR_{Water} is the intake rate (ingestion of water) (730 L/y); and
DCF_{Ing} is the internal dose conversion factor for ingestion (rem/μCi).

The $PDCF_{Water}$ is the product of the intake rate (IR_{Water}) and the nuclide-specific dose conversion factor (DCF_{Ing}) (i.e., the denominator in Equation 5-23). Therefore, the nuclide-specific $PDCF_{Water}$ will be the same for each site. The values of $PDCF_{Water}$ for drinking water ingestion are presented in Table 5-6.

Table 5-6. Dose Conversion Factors for the Water Pathway (EPA, 1988a unless otherwise noted)

Radionuclide	PDCF Ingestion [a] (rem/y)/(μCi/L)
H-3	4.67E-02
C-14	1.52E+00
Al-26	1.06E+01
Si-32	7.99E+00
Cl-36	2.21E+00
K-40	1.36E+01
Co-60	1.97E+01
Ni-59	1.53E-01
Ni-63	4.21E-01
Se-79	6.35E+00
Sr-90	1.12E+02
Zr-93	1.59E+00
Nb-93m	3.81E-01
Nb-94	5.21E+00
Tc-99	1.07E+00
Pd-107	1.09E-01
Ag-108m	5.56E+00
Cd-113m	1.17E+02
Sn-121m	1.28E+00
Sn-126	1.53E+01
I-129	2.01E+02
Cs-135	5.16E+00
Cs-137	3.65E+01
Ba-133	2.48E+00
Sm-151	2.84E-01
Eu-152	4.73E+00
Eu-154	6.97E+00
Pb-210	5.30E+03
Ra-226	6.27E+03
Ra-228	1.64E+03

Radionuclide	PDCF Ingestion [a] (rem/y)/(μCi/L)
Th-229	2.94E+03
Th-230	b
Th-232	3.63E+03
Pa-231	1.85E+04
U-232	1.55E+03
U-233	b
U-234	b
U-235	b
U-236	1.96E+02
U-238	2.07E+02
Np-237	3.24E+03
Pu-238	2.34E+03
Pu-239	2.58E+03
Pu-240	2.58E+03
Pu-241	4.82E+08
Pu-242	2.45E+03
Pu-244	4.11E+03
Am-241	2.66E+03
Am-243	2.64E+03
Cm-243	1.83E+03
Cm-244	1.47E+03
Cm-245	4.56E+03
Cm-246	2.70E+03
Cm-247	2.70E+03
Cm-248	9.94E+03
Cf-249	3.46E+03
Cf-250	2.64E+08
Cf-251	3.54E+03

[a] PDCF is based on a water consumption rate of 2 L per day.

[b] PDCF is based on curve-fit equations for PDCFs (see Appendix E of Volume 2):

Th-230 PDCF = $343.6 + 2.644*T - 4.9E\text{-}4*T^2 + 4.274E\text{-}08*T^3 - 1.44E\text{-}12*T^4$

U-233 PDCF = $217.3 + 0.26013*T - 8.656E\text{-}6*T^2$

U-234 PDCF = $206.2 + 5.612E\text{-}3*T + 8.032E\text{-}06*T^2 - 4.175E\text{-}10*T^3$

U-235 PDCF = $188.6 + 0.3909*T - 3.728E\text{-}06*T^2$

These curve fit equations are for arrival times of the radionuclides at the performance boundary between 100 and 10,000 y. If arrival time exceeds 10,000 y, the calculation is made for T=10,000 y.

The performance measure for atmospheric releases is based on all exposure pathways, and the associated dose conversion factors are associated with ingestion of vegetables, meat, and milk; inhalation of radionuclides; and, if applicable, external exposure. The performance measure for exposure of inadvertent intruders is based on several exposure pathways (e.g., direct external exposure to waste, ingestion of food or soil, and inhalation of airborne suspended soil particles). Factors that are used to calculate the doses and therefore the scenario dose conversion factors include correction factors for the amount of natural soil mixed with exhumed waste, plant-to-soil concentration ratios, shielding factors, and the fraction of time the individual is exposed through a certain pathway. Details of the approach are given in Appendices C and D of this volume.

5.4.2 Treatment of Radioactive Decay Products

Buildup and decay of all radiologically significant decay products of radionuclides in the waste inventory at the time of disposal are taken into account in the PE, based on use of the Bateman equations to estimate the activity of the decay products at any time after disposal. However, particularly for transport in groundwater (which is a relatively complex problem for radioactive decay chains with long-lived decay products), simplifying assumptions are used to estimate the contributions from the decay products. These assumptions should result in conservative (i.e., high) estimates of dose per unit activity of the parent radionuclide in the disposal facility.

Many radionuclides that are assumed to be present in MLLW at DOE sites decay to products that are also radioactive. Although the list of radionuclides in the assumed inventory does not explicitly include radioactive decay products that are not present initially, all decay products are included in the PE when they could contribute significantly to the dose. Following this approach, contributions to dose from buildup and decay of the radioactive decay products are individually considered for the water pathway following releases to groundwater or surface waters, for exposures of off-site individuals following airborne releases of radionuclides, and for exposures of inadvertent intruders.

Three different situations involving radioactive decay products must be considered in the PE. The first is the common occurrence of radioactive decay products that are short-lived compared with both the half-life of the parent radionuclide and the time after disposal at which exposures are assumed to occur. Examples of this situation include Y-90 produced in the decay of Sr-90, and Ra-228 and its shorter-lived decay products produced in the decay of Th-232. For all radionuclides with such short-lived decay products, the dose for any exposure situation of concern is estimated by assuming that the activity of the decay products is in equilibrium with the activity of the parent at the assumed exposure location, taking into account the appropriate branching fractions in the decay of the parent.

The second situation is the frequent occurrence in decay chains for long-lived actinides of a long-lived parent radionuclide that decays to a radionuclide or radionuclides that are long-lived, but still shorter-lived than the parent. Examples of this situation include U-234, Th-230, and Ra-226 produced in the decay of U-238; and Pa-231 produced in the decay of U-235. In such decay chains, the half-lives of the decay products are sufficiently long that their activity usually will not be in equilibrium with the activity of the parent at the time exposures are assumed to occur.

For transport of radionuclides by the groundwater pathway, a proper treatment of the type of decay chain described above is a rather complex problem involving convolution integrals in the form of Laplace transforms, even when a one-dimensional flow model without dispersion is assumed and transport of radionuclides is determined by the distribution coefficient, K_d (Rosinger and Tremaine, 1980). Therefore, a simplified approach is used in the PE. The concentration of the parent radionuclide in water at the performance boundary and the time after disposal at which the radionuclide reaches the performance boundary are calculated based essentially on the maximum annual release from the disposal facility, time of the release, and the retarded transport time of the radionuclide to the performance boundary. From these results, the concentrations of the decay products in water at the performance boundary are calculated using the Bateman equations for radioactive decay. This approach assumes, in effect, that all decay products are produced at the performance boundary itself, rather than along the entire path of travel. Because the parent radionuclide and decay products generally have different sorption characteristics and would generally not travel together, this approach should give a conservative (i.e., higher) estimate of possible contributions of long-lived decay products to the dose from the water pathway.

For releases of radionuclides to air and exposures of inadvertent intruders to solid waste, time delays between releases and subsequent exposures of the kind described above for the groundwater pathway would not be of concern. Therefore, for long-lived radionuclides that decay to long-lived products that are shorter-lived than the parent, the Bateman equations are used to estimate the activity of the decay products at the time releases or exposures are assumed to occur, and these activities then are used to estimate the dose from a parent radionuclide and its decay products.

The third situation involves a decay product that is longer-lived than its parent radionuclide. Examples of this situation include Am-241 produced in the decay of Pu-241 and U-234 produced in the decay of Pu-238. When a decay product is longer-lived than its parent, the maximum activity of the decay product is approximately equal to the initial activity of the parent multiplied by the ratio of the half-lives of the parent and the decay product, but there is never any kind of equilibrium between the activities of the parent and decay product.

In principle, the dose from a radionuclide and its longer-lived decay product for any exposure situation can be estimated by using the Bateman equations to obtain the activity of the decay product at the time exposures are assumed to occur. However, if exposures are expected to occur at a time sufficiently long after disposal that the activity of the shorter-lived parent is reduced to innocuous levels, as might be expected to occur in groundwater transport of relatively short-lived transuranic radionuclides (e.g., Pu-238 and Pu-241), a simple and conservative approach would be to convert the initial activity of the parent to an equivalent initial activity of the decay product using the ratio of the two half-lives and then using the activity of the decay product as input to the transport and exposure analysis. This is the approach used in the PE.

Additional details about the treatment of radioactive decay products in the PE are presented in Appendix E of this volume.

6. CONCEPTUAL MODEL ASSUMPTIONS AND PARAMETER SENSITIVITY ANALYSES

This chapter presents a discussion of the major assumptions for the conceptual models used in the PE and provides a brief parameter sensitivity analysis for the models used in the performance evaluation. The parameters shown to be of minor importance are discussed and dismissed in this section. The effects of the most important parameters on site-specific results are discussed in each of the site chapters contained in Volume 3 of this report.

6.1 DISCUSSION OF MAJOR ASSUMPTIONS IN CONCEPTUAL MODELS

The major simplifying assumptions used in the conceptual models for the water and atmospheric pathways and the intruder scenarios are presented in Chapter 5 of this volume. The effect of these assumptions on the results of the analysis are discussed in this section. Several assumptions relate to the conceptual models for all pathways and scenarios, so they are discussed separately under "All Pathways and Scenarios."

The PE was developed for use as a screening tool. As such, those responsible for this effort used many simplifying assumptions to develop the conceptual models for the water and atmospheric transport and intrusion scenarios. Most of these simplifying assumptions tended to provide conservatism. For purposes of this discussion, conservative means that the maximum permissible waste concentrations calculated by the PE method are likely to be lower than if a more detailed analysis were done, such as in a performance assessment. Parameters within each of the pathway models for which major assumptions were made are discussed in this section and are listed in Table 6-1.

Table 6-1. Major Parameters for Which Simplifying Assumptions Were Made in Developing the PE Conceptual Models

Pathway	Parameter
All Pathways and Scenarios	Waste form and performance
	Performance of engineered barriers
	Radionuclide combinations
	Applicable regulations
Water	Solubility constraints
	Continuous source
	Sorption effects
	Fractured flow
	Regional recharge
Atmospheric	Volatile radionuclide transport
	Volatility and chemical forms
Intrusion Scenario	Applicable scenarios
	Time of intrusion

Performance assessments have been developed for LLW disposal facilities at some sites evaluated by this project (INEL, Hanford, ORR, and SRS). For these sites, comparisons have been made between the results of the PEs and the site-specific analyses. These are presented in Appendix A of Volume 3 and are summarized in the appropriate site chapters in Volume 3. Because the differences in results between the site-specific performance assessments and the PEs are largely due to differences in conceptual models, the differences in conceptual models are also described in Appendix A. At the time the PE was being performed, performance assessments were being drafted for LLW disposal facilities at NTS and LANL; available information from the draft performance assessments was incorporated in the appropriate site chapters in Volume 3.

6.1.1 All Pathways and Scenarios

Several of the assumptions used in the PE were basic to all pathways. These pertain to the (1) waste form and performance, (2) performance of engineered barriers, (3) radionuclide combinations, and (4) applicable regulations. Included in this section is a discussion of the effects of these assumptions on the three primary exposure pathways.

6.1.1.1 Waste Form and Performance

Grout is the waste form evaluated in the PE because the majority of treated and stabilized DOE MLLW is anticipated to be stabilized by this method, although other waste forms may also be used. A less-engineered waste form likely to be disposed of as MLLW is debris and demolition wastes, which may only receive some type of prior surface decontamination. A vitrified waste form has also been proposed for stabilizing LLW and would represent a more stable waste form compared to grout; however, due to the large costs of vitrification and the early stage of development of this technology, it is not likely to be widely used for MLLW currently. The degree of waste form performance is site specific and waste specific, and some sites may not require a more highly engineered waste form.

Water Pathway. With respect to the water pathway, the performance of the waste form used in the PE is encompassed in the source concentration reduction factor, CRF_{Source}, which is defined as the ratio of the radionuclide concentration in the solid waste to the resulting concentration in the leachate. As mentioned in Section 5.3.1, solubility constraints are not considered. The effect of this assumption on the CRF_{Source} is discussed later in this section. The PE was designed to allow easy substitution of waste form models and any waste form performance model can be substituted for the CRF_{Source}. Many mechanisms can be used to represent the distribution of the radionuclides between the grouted waste and the surrounding leachate including solubility limits, equilibrium desorption, kinetic desorption, and diffusion. The model used in the PE for the grouted waste form is equilibrium desorption with infiltrating water through the disposal facility.

The combination of desorption with infiltration is commonly used as a source-term model for grouted waste in performance assessments for LLW disposal. When this release scenario is considered, diffusion is usually primarily important for early, low-rate releases at a time when infiltration rates through the disposal facility are low. When infiltration rates increase, desorption and transport with the infiltrating waters through the facility dominate the diffusion transport.

The effect of neglecting diffusion as a transport mechanism is considered minor for the purposes of this analysis. With respect to source term, the PE calculations estimate only the peak release from the disposal facility, and the effect of diffusion on the peak release rate changes the timing but not the magnitude.

With respect to the kinetic desorption approach, equilibrium desorption is conservative because equilibrium between radionuclides in the solid and liquid phases is assumed to occur instantaneously. This results in higher leachate concentrations.

Using alternative waste forms (e.g., vitrified debris) could have a much larger impact on the results of the analysis than modifying the release mechanisms for grouted waste. For example, leachate from a vitrified waste form would likely be modeled as a dissolution process that is dependent on the chemical equilibrium between the silicate minerals encapsulating the waste and the surrounding leachate solution. Due to the much lower solubility of the accompanying radionuclides, the CRF_{Source} for a vitrified waste could be several orders of magnitude larger than those used for a grouted waste form. The process of vitrification may remove many of the more volatile radionuclides, a situation where a reduced risk to the public from disposal may be offset by an increased risk to workers from treatment.

A debris waste form would likely be modeled as a desorption or rinsing process which would probably produce a CRF_{Source} of lower magnitude than that provided by the grout model.

Atmospheric Pathway. The conceptual model for the atmospheric pathway assumes that the waste form has the physical properties of the surrounding soils with respect to diffusion of volatile radionuclides into the atmosphere. This assumption may be reasonable for a grout or debris waste form but may not be appropriate for a vitrified waste form. A vitrified waste form disposed in the shallow subsurface would have a very low release rate, if any, with respect to atmospheric releases, one of the primary reasons being, as previously pointed out, that the more volatile radionuclides may have been removed during vitrification.

Intruder Scenarios. The intruder scenarios assume that the waste form is indistinguishable from soil at the time of intrusion. This assumption may be reasonable for a grout or debris waste form but may not be appropriate for a vitrified waste form. A vitrified waste form disposed in the shallow subsurface could be expected to maintain its structural integrity for a much longer time than a grout or debris waste form. The assumed intruder scenarios may be reasonable for this type of waste, but the transfer of contaminants would be substantially less to none.

In summary, the choice of waste form will significantly affect the PE estimates of the permissible concentrations of radionuclides in the waste for the water and atmospheric pathways and the intruder scenarios. The grouted waste form was used in this analysis because it is expected to be a common waste form for MLLW disposal. However, the main effect of a vitrified waste form would be to increase the permissible concentrations of radionuclides in the waste. The grouted form may be reasonably comparable to a debris waste form.

6.1.1.2 Performance of Engineered Barriers

For the water pathway, the engineered barriers in the PE are assumed to provide waste isolation for at least 100 y for the generic trench and at least 300 y for the generic tumulus. As demonstrated in Table 7-2 in this volume, the difference in performance of the two facilities affects only the shorter-lived radionuclides (e.g., H-3), with the tumulus design providing greater permissible waste concentrations. For longer-lived radionuclides (e.g., Tc-99), the difference in the estimated permissible concentrations of radionuclides in the waste for the two facilities is negligible.

Appendix A of Volume 3 presents a discussion of the performance of the site-specific, above-ground vault design (the Mixed Waste Disposal Facility [MWDF]) planned for the Savannah River Site (SRS). The performance of the engineered barriers for this disposal facility was assumed to be 3000 y in the site-specific performance assessment for a comparable LLW facility (MMES, 1994). For the water pathway, comparison of the permissible waste concentrations calculated for the MWDF with those for the tumulus in Table 7-4 of this volume for the SRS shows that the Tc-99 concentrations are within a factor of three, while H-3 concentrations are not limited by the MWDF, which indicates that the longer duration engineered barriers provide significantly higher permissible concentrations of radionuclides in the waste for the shorter-lived radionuclides. Differences in permissible waste concentrations for longer-lived radionuclides are minor.

The assumed performance of engineered barriers for the water and atmospheric pathways also affects the PE estimates of the permissible concentrations of radionuclides in the waste for the intruder scenarios. In the PE, the time of a homesteader intrusion is assumed to occur 300 y after closure of the trench facility and 500 y after closure of the tumulus facility, and the time of a post-drilling intrusion is assumed to occur as early as 100 y. For the same reasons as discussed for the water pathway, the primary effect of these time-of-intrusion assumptions pertains to shorter-lived radionuclides. The engineered barriers are designed for long-term performance, but the performance cannot be assured. Based on experience, performance of engineered barriers is hard to justify for more than a few hundred to a few thousand years; few engineering containment projects have been designed to last thousands of years. While these barriers may remain intact for thousands of years, allowing credit for only a few hundred years duration may introduce conservatism into the analysis.

In general, two basic principles apply to the disposal of radioactive waste. The first is to contain the waste while it decays to acceptable levels. This principle applies to shorter-lived radionuclides that can be contained by natural or engineered structures that can reasonably be expected to remain intact for a few hundred years. For longer-lived radionuclides, complete containment is difficult to demonstrate with any certainty. This situation requires use of the second principle, which is to control the release rate so that potential individual exposures from those releases will not be excessive. Slow release implies that the containment features degrade slowly, with no abrupt failures that might lead to large increases in the projected doses to exposed individuals.

To provide consistency of analysis and to emphasize the differences in natural conditions among the sites, the PE team assumed that two types of disposal facilities, a generic trench and a generic tumulus, were to be used. Performance of these facilities has been modeled in a simplified and conservative manner based on the approaches used in many LLW performance assessments. As illustrated by the MWDF at SRS, sites that require additional long-term performance of engineered barriers can achieve this objective by additional attention to more robust design of the disposal facility.

6.1.1.3 Radionuclide Combinations

The PE calculations provided estimated maximum permissible concentrations for individual radionuclides by assuming that each individual radionuclide provides the entire permissible dose. This approach allows the flexibility to evaluate the acceptability of various combinations of radionuclide inventories. One of the effects of this approach, however, is that estimates of the permissible concentrations for individual radionuclides in the waste are higher than would be permitted for a combination of radionuclides. This approach has been taken because the actual radionuclide inventories at each evaluated site are unknown.

To evaluate the acceptability of disposal of various combinations of MLLW waste streams, the sum of fractions rule is applied. Using this approach, the acceptability of a multiple-radionuclide inventory is determined by summing the ratios of each radionuclide waste concentration to its permissible concentration. A sum less than or equal to one indicates that the inventory is acceptable for disposal.

Some site-specific performance assessments are based on radionuclide inventories and volumes, not on the maximum permissible concentrations for individual radionuclides in the waste. More recent performance assessments use an approach similar to that used in the PE. To permit direct comparisons of the results of the PAs and the PEs, the analyses of the PAs presented in Appendix A of Volume 3 are based on maximum permissible concentrations of the individual radionuclides.

6.1.1.4 Applicable Regulations

The performance measures used in the PE are 4 mrem (0.04 mSv) per year for exposures from the water pathway and 10 mrem (0.10 mSv) per year for the atmospheric exposure pathway. Both of these limits have been questioned, and depending on circumstances, it is possible that the limits could be changed in the future.

Should a new regulatory limit be set in the future that changes any pathway performance objective, the PE results could be modified by multiplying the affected pathway waste concentration by the ratio of the new and old performance objective. For example, should a new limit for the water pathway be set at 15 mrem per year, radionuclide waste concentrations for the water pathway would be increased by the ratio (15 mrem per year/4 mrem per year), that is, the permissible concentrations for radionuclides in the waste, as controlled by the water pathway, would increase by a factor of almost four. Other changes in regulations could be treated similarly. The PE method presented here contains sufficient flexibility to readily accommodate such changes.

6.1.2 Water Pathway

Five major parameters for which assumptions have been made are discussed in this section: (1) solubility constraints, (2) continuous source, (3) sorption effects, (4) fractured flow, and (5) regional recharge. These assumptions tend to increase the conservatism of the PE analysis. Each of these assumptions is discussed below.

6.1.2.1 Solubility Constraints

As a simplifying assumption, the PE calculations do not include the effects of solubility limits in the estimates of permissible waste concentrations, even though it is recognized that such limits can potentially increase the permissible waste concentration dramatically. This assumption implies that all radionuclide concentrations are inventory limited, and that there is a direct, linear relationship between the permissible dose at the performance boundary and the permissible waste concentration.

In cases where the calculated leachate concentration (i.e., the highest dissolved concentration) for a radionuclide is higher than its maximum solubility, then the radionuclide is solubility limited and not inventory limited, effectively resulting in no limit on permissible waste concentration for the water pathway using the methods developed for the PE. For example, a radionuclide with a high concentration in disposed waste and a low solubility can have a maximum leachate concentration no higher than its solubility limit. Should this leachate concentration be sufficiently low that mixing of leachate with groundwater provides a groundwater radionuclide concentration equivalent to less than 4 mrem per year, then the radionuclide is solubility limited and there will be no restriction on its concentration in the disposed waste based on the water pathway.

However, when other radionuclides are present in the disposal facility (as is the case for actual MLLW), the concentration of each individual radionuclide will be lower than if the estimated permissible concentration were based on each radionuclide contributing the entire 4 mrem annual dose. The sum-of-fractions rule (see Section 6.1.1.3) is used to estimate the permissible concentrations for multiple radionuclides in MLLW. To use the sum-of-fractions rule, an estimate of the maximum permissible radionuclide concentration is required, even if that maximum is higher than allowed by solubility limits. An indication of the degree to which the solubility limit is exceeded by eight indicator radionuclides is presented in this section.

The highest aqueous concentrations estimated in the PE are in the leachate exiting the disposal facility. (The PE leachate concentration can be determined by dividing the permissible waste concentration by the CRF_{Source} and the 1000 L/m^3 conversion factor.) The highest leachate concentrations from all 15 sites for eight indicator radionuclides (identified and discussed in Chapter 7 of this volume) are compared in Table 6-2 with solubility limits compiled from the literature (ORNL, 1994, Appendix C). This comparison shows that, for three of the radionuclides (Cs-137, U-238, and Pu-239), the solubility limit is less than the estimated leachate concentration; for four (C-14, Sr-90, Tc-99, and Am-241), the estimated leachate concentration is less than the solubility limit.

Table 6-2. Comparison of the Highest Leachate Concentrations from the Generic Tumulus for the 15 Sites Evaluated in the PE with Solubility Limits Compiled from the Literature (limits are for the most soluble species) (ORNL, 1994, Appendix C)

Indicator Radionuclide	Site with Highest Permissible Waste Concentration	PE Leachate Concentration (μCi/L)	Literature Solubility Limit (μCi/L)	Chemical Formula for Literature Solubility Limit
H-3	LLNL/SNL/LANL/ Pantex/WVDP	NL[a]	--[b]	T_2O
C-14	LANL	2E4	1E5	$BaCO_3$
Sr-90	Paducah	7E4	2E6	$SrCO_3$
Tc-99	Pantex/LANL	1E0	3E5	NH_4TcO_4
Cs-137	All Sites	NL	5E11	Cs_2CO_3
U-238	LANL/INEL/Pantex	5E-3	1E-4	UO_2
Pu-239	ANLE	7E-1	3E-1	PuO_2
Am-241	Pantex	2E0	6E3	Am_2O_3

a No Limit - the PE leachate concentration is above the specific activity of the pure, elemental radionuclide.
b A solubility limit for tritium is not realistic for this analysis because the form of tritium is assumed to be tritiated water.
No water pathway analysis was conducted for NTS.

For Cs-137, the inventory is not limited even without considering solubility limits, so that application of solubility limits does not provide higher inventory limits. For U-238, the solubility limit is about fifty times smaller than the highest leachate concentration, so including the solubility limit would provide higher permissible waste concentrations for this radionuclide. The maximum leachate concentration for Pu-239 is slightly higher than the literature solubility limit value, indicating that Pu-239 would be solubility limited. This would result in a higher permissible concentration for this radionuclide in the waste. Neglecting solubility limits will have no impact for radionuclides with unlimited permissible waste concentrations. These results indicate that neglecting solubility limits does not have a significant effect on the results except for U-238.

The use of solubility limits is complicated by many factors, including (1) the assumed chemical speciation of the element, (2) interactions of multiple elements, and (3) conditions of pH and Eh and the aqueous environment as it changes from a high-pH grouted waste form to a more pH-neutral groundwater. Each of these factors is discussed briefly below.

Wide ranges of solubility limits are found for many radionuclides depending on the assumed chemical speciation. The exact chemical speciation of the radionuclide is generally not known after disposal and therefore must be assumed. Therefore, the uncertainty associated with solubility limits due to the assumed chemical form of the radionuclides can be very large. Neglecting solubility limits is analogous to assuming a very high chemical solubility; such an approach tends to maximize leachate concentrations.

Selection of solubility values is hampered by the lack of knowledge of environmental conditions, particularly pH and Eh, especially considering the previously cited large changes in conditions between a grouted waste and natural waters at a site. Solubility is a function of these parameters, and the lack of knowledge of long-term, site-specific conditions makes selection of specific solubility limits difficult.

6.1.2.2 Continuous Source

A continuous source of radionuclides from each disposal facility was assumed in the PE because, although the duration of release depends on the radionuclide inventory, the site-specific waste inventories are not known. Because of the shape of a contaminant plume resulting from a continuous source, the primary consequence of this assumption is that longitudinal dispersion is neglected in the water pathway calculations. Another effect of the continuous source assumption relates to sorption modeling and is discussed in the next section. As discussed in Appendix B.3 of this volume, longitudinal dispersion affects the shape of the contamination front for a continuous source and provides some peak attenuation for a finite source. The dispersion coefficient can be represented as being proportional to pore-water velocity. As discussed in Appendix A.1 of Volume 3, for the INEL site, which has one of the largest pore-water velocities of the 15 sites, including the attenuation effects of dispersion increases the permissible waste concentrations for the water pathway by about 20% when compared with the no-dispersion case.

The continuous source assumption can also affect attenuation in the subsurface. Calculations for the LLW performance assessment at Oak Ridge (ORNL, 1994) were based on the assumption that sorption effects in the vadose zone attenuated the dissolved radionuclide concentrations in the subsurface. As discussed in Appendix A.3 of Volume 3 of this report, this attenuation effect is related to the duration of the source from the facility, with shorter source durations providing larger attenuation effects. The effect is more pronounced for radionuclides with higher K_d values. A continuous source assumption results in a more conservative (i.e., lower) permissible waste concentration with respect to the models used in the Oak Ridge performance assessment.

6.1.2.3 Sorption Effects

Sorption is treated as a linear and reversible equilibrium process in the PE, which results in retardation of the radionuclides but no concentration attenuation due to the assumption of a continuous source. Other approaches have been used in LLW performance assessments which include processes (e.g., partially irreversible sorption) that cause concentration attenuation during transport. Calculations for the LLW performance assessment at Oak Ridge were based on a vadose zone sorption model which provided concentration attenuation proportional to the assumed partition coefficient and duration of release. Calculations for the Hanford performance assessment of the 200 West Area (Wood et al., 1994b) were based on a sorption model which provided concentration attenuation proportional to the retardation factor, R, with larger R values resulting in increased concentration attenuation. The sorption model used in the PE provides more conservative (i.e., lower) permissible waste concentrations than these two models by as much as two orders of magnitude (see Appendix A.4 in Volume 3).

A wide variety of sorption models has been developed. However, most of these require a considerable amount of knowledge of the nature of the sorbent, sorbate, and the solution chemistry, which is beyond the scope of the PE. The PE incorporates a simple and widely used approach consisting of a linear model utilizing a distribution coefficient (K_d). As discussed by Galya (1987), when dispersive effects are considered for a finite-duration source, sorption accentuates the dispersive concentration attenuation by slowing contaminant movement and providing more time for the dispersive effects to act.

6.1.2.4 Treatment of Fracture Flow

Fractured geologic material is present at a few of the sites evaluated in the PE project. Flow in these materials was treated in the PE in the following manner. When fractured materials are present in the vadose zone, flow through these materials is assumed to be complete and instantaneous relative to the matrix flow in the non-fractured geologic materials. This approach is consistent with the performance assessment analysis for INEL LLW. Not considering sections of the vadose zone due to the presence of fractures is usually conservative because the fractures generally do not transmit water except during extreme events such as floods, and the disposal facilities are assumed to be located out of the floodplains. Because of capillary effects, only the smaller pores within the rock matrix will contain water; the fractures will contain air.

When fractured materials are present in the saturated zone, the fractures and matrix porosity are treated through application of an assumed combined effective porosity and a larger hydraulic conductivity. Pore velocity and contaminant travel times are based on this effective porosity and hydraulic conductivity. This approach is a common way of treating regional fractured groundwater flow.

6.1.2.5 Regional Recharge

In the PE, the concentration in the saturated zone resulting from dilution of leachate with groundwater is assumed to be constant during transport to the 100-m performance boundary. As shown in Section A.3 of Volume 3 of this report, for ORR additional concentration dilution is provided by mixing of infiltrating water from regional recharge with the contaminated groundwater. While this effect is usually quite small, Oak Ridge dilution due to regional recharge provides more attenuation than dilution of leachate with groundwater. Therefore, sites with high recharge relative to groundwater flow and low CRF_{Water} values would benefit most from inclusion of dilution from regional recharge.

6.1.3 Atmospheric Release

Assumptions related to two major parameters in the conceptual model for the atmospheric pathway are discussed in this section: (1) volatile radionuclide transport and (2) volatility and chemical forms.

6.1.3.1 Volatile Radionuclide Transport

Transport of radionuclides from a disposal facility to the soil surface can occur through several potential processes including gas and vapor diffusion, desiccation cracks, erosion, plant root uptake, and burrowing animals. Transport via several of these potential pathways has received relatively little research, and useful transport models are generally not available. The gas- and vapor-phase diffusion model was chosen to represent the potential transport of volatile radionuclides because these processes are recognized as important transport mechanisms and data are available to perform the analysis. The data chosen for the diffusion analysis were intentionally conservative (i.e., tended to increase the transport rate) to attempt to capture the uncertainty of representing other possible transport mechanisms.

The conceptual model for the atmospheric pathway is based on transport through soils with porosity and moisture content similar to that occurring naturally at the site. In practice, however, it is almost certain that a MWDF will have a cover and liner that will include a low permeability vapor barrier to limit emanation of volatile contaminants. Vapor barriers generally consist of low permeability soil layers that are constructed to have low porosity and small interstitial pore diameters. Because select materials are used, it is not likely that they will ever have characteristics similar to native soils. Therefore, the assumption of natural soil conditions provides further conservatism for the atmospheric pathway.

6.1.3.2 Volatility and Chemical Forms

Implicit in the use of a gas- or vapor-phase diffusion model is the assumption that the radionuclides of interest will be volatile under disposal facility conditions. All wastes are assumed to be disposed of in a stabilized and solidified form containing no pressurized gases. Therefore, the transition from solid to gas or vapor form must occur prior to gas or vapor-phase transport.

Tritium (H-3) and C-14 are the only two radionuclides considered in the atmospheric pathway analysis. Other radionuclides, including I-129 and Cs-137, become volatile under high temperature conditions but are not expected to be volatile under disposal facility conditions. The disposed H-3 is assumed to become associated with the water (substituting for a normal hydrogen atom) and to mix with the water passing through the disposal facility, with the associated water vapor providing the transport mechanism for the atmospheric transport. The radionuclide C-14 is assumed to be transformed to carbon dioxide gas (CO_2), which provides the mechanism for atmospheric transport. Each of these assumptions is discussed below.

Tritium will likely be present in the disposed waste as a form of hydride (e.g., lithium hydride) or as tritiated water (^3HHO). Tritium is then assumed to become associated with the water infiltrating through the disposal facility and is assumed to be completely present in the water phase. The water vapor present in the disposal facility will then contain tritium and the vapor migration to the ground surface provides the pathway assumed in the PE. As the relative humidity decreases, the driving force for evaporation increases, suggesting that arid disposal sites are more likely to have a larger tritium transport in the vapor phase than more humid sites.

Carbon compounds are converted to a gas (CO_2) in disposal sites such as municipal landfills by microbial degradation of organic materials. This mechanism for carbon dioxide generation is implicit in the PE with regard to C-14. To occur, biodegradation requires the presence of several factors. These include an electron donor (e.g., organic carbon), terminal electron acceptors, adequate nutrients, an acceptable pH, proper temperature, absence of toxic constituents, and adequate mixing, time, and moisture. Several factors related to the waste form used in the PE make the presence of some of these conditions doubtful. The high pH associated with grout (greater than 12) would dramatically slow the biodegradation process. Should carbon dioxide be generated, the high pH will cause it to be transformed to highly soluble carbonate and bicarbonate ions. These factors are not explicitly considered in the PE analysis for atmospheric transport.

6.1.4 Intrusion Scenarios

The homesteader and post-drilling scenarios used in the PE are the same for all sites except for SRS which has a later time of intrusion for the post-drilling scenario. Although future social behavior, including intrusion scenarios, is difficult to predict, these two scenarios were used because, based on experience gained in performance assessments (ORNL, 1994; MMES, 1994), they tend to be the most restrictive of the standard scenarios evaluated. Particularly, the scenarios that resulted in long-term, chronic exposure (e.g., homesteader and post-drilling) have been shown to be more restrictive than scenarios based on a one-time, acute exposure (e.g., discovery, drilling operations). At many sites, particularly in the arid regions of the country, the intrusion scenarios provide the most restrictive permissible waste concentrations for most radionuclides.

The effects of the assumptions for two major parameters related to the selection of intrusion scenarios are discussed here: (1) applicable scenarios and (2) time of intrusion.

6.1.4.1 Applicable Scenarios

At each of the 15 sites, the lesser of the permissible radionuclide concentrations in the waste calculated from the homesteader and post-drilling scenarios is compared to the comparable estimates for the water and atmospheric analyses. Under some conditions, the homesteader intrusion scenario may not be credible. For example, when the waste is disposed below grade at a sufficient depth to preclude intrusion for the construction of a basement, the homesteader scenario becomes unlikely. This situation can occur only at sites with sufficiently thick vadose zones to allow disposal at such depths, an attribute of the more arid sites, and would generally be more costly than shallow disposal.

Assuming that the wastes can be disposed of at a depth which precludes inadvertent human intrusion under the homesteader scenario, the permissible waste concentrations for intrusion will be based solely on the post-drilling scenario. Table 6-3 shows the permissible waste concentrations for the generic trench and tumulus designs based on the homesteader and post-drilling scenarios. The most restrictive waste concentrations are shown in bold italics. Analysis of the data in Table 6-3 shows that 47 and 43 of the 58 radionuclide concentrations for the generic trench and tumulus designs, respectively, are limited by the homesteader scenario.

Table 6-3. Permissible Waste Concentrations for the Standard Intrusion Scenarios (most limiting radionuclide concentrations for each facility design is highlighted in bold italic) (Part 1 of 2)

Nuclide	Trench		Tumulus		Nuclide
	$C_{W\text{-}Intr}$ Homesteader [a] ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Post-Drilling ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Homesteader [a] ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Post-Drilling ($\mu Ci/m^3$)	
H-3	1E+12	***7E+07***	NL	***7E+07***	H-3
C-14	***1E+04***	7E+04	***1E+04***	7E+04	C-14
Al-26	***5E+01***	5E+04	***5E+01***	5E+04	Al-26
Si-32	***9E+03***	1E+04	4E+04	***1E+04***	Si-32
Cl-36	***2E+02***	9E+02	***2E+02***	9E+02	Cl-36
K-40	***7E+02***	2E+04	***7E+02***	2E+04	K-40
Co-60	NL	***3E+10***	NL	***3E+10***	Co-60
Ni-59	***3E+06***	1E+07	***3E+06***	1E+07	Ni-59
Ni-63	***8E+06***	1E+07	3E+07	***1E+07***	Ni-63
Se-79	***2E+05***	8E+05	***2E+05***	8E+05	Se-79
Sr-90	1E+06	***5E+04***	1E+08	***5E+04***	Sr-90
Zr-93	***5E+06***	3E+07	***5E+06***	3E+07	Zr-93
Nb-93m	5E+12	***4E+09***	NL	***4E+09***	Nb-93m
Nb-94	***9E+01***	9E+04	***1E+02***	9E+04	Nb-94
Tc-99	***2E+04***	8E+04	***2E+04***	8E+04	Tc-99
Pd-107	***6E+06***	3E+07	***6E+06***	3E+07	Pd-107
Ag-108m	***5E+02***	1E+05	***1E+03***	1E+05	Ag-108m
Cd-113m	4E+09	***1E+06***	7E+13	***1E+06***	Cd-113m
Sn-121m	***2E+07***	5E+07	2E+08	***5E+07***	Sn-121m
Sn-126	***8E+01***	7E+04	***8E+01***	7E+04	Sn-126
I-129	***2E+03***	1E+04	***2E+03***	1E+04	I-129
Cs-135	***2E+05***	8E+05	***2E+05***	8E+05	Cs-135
Cs-137	***3E+05***	8E+05	3E+07	***8E+05***	Cs-137
Ba-133	2E+11	***3E+08***	NL	***3E+08***	Ba-133
Sm-151	***2E+08***	***2E+08***	8E+08	***2E+08***	Sm-151
Eu-152	6E+08	***2E+07***	2E+13	***2E+07***	Eu-152
Eu-154	4E+12	***4E+08***	NL	***4E+08***	Eu-154
Pb-210	7E+06	***7E+04***	4E+09	***7E+04***	Pb-210
Ra-226	***8E+01***	3E+03	***9E+01***	3E+03	Ra-226
Ra-228	NL	***4E+09***	NL	***4E+09***	Ra-228
Th-229	***5E+02***	3E+04	***5E+02***	3E+04	Th-229
Th-230	***8E+01***	7E+04	***8E+01***	7E+04	Th-230
Th-232	***6E+01***	2E+04	***6E+01***	2E+04	Th-232
Pa-231	***2E+02***	5E+03	***2E+02***	5E+03	Pa-231

Table 6-3. Permissible Waste Concentrations for the Standard Intrusion Scenarios (most limiting radionuclide concentrations for each facility design is highlighted in bold italic) (Part 2 of 2)

Nuclide	Trench		Tumulus		Nuclide
	$C_{W\text{-}Intr}$ Homesteader [a] ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Post-Drilling ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Homesteader [a] ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Post-Drilling ($\mu Ci/m^3$)	
U-232	***2E+03***	5E+04	***1E+04***	5E+04	U-232
U-233	***7E+02***	1E+05	***7E+02***	1E+05	U-233
U-234	***1E+03***	1E+05	***1E+03***	1E+05	U-234
U-235	***6E+02***	1E+05	***6E+02***	1E+05	U-235
U-236	***2E+04***	1E+05	***2E+04***	1E+05	U-236
U-238	***5E+03***	1E+05	***5E+03***	1E+05	U-238
Np-237	***4E+02***	4E+03	***4E+02***	4E+03	Np-237
Pu-238	***7E+04***	1E+05	4E+05	***1E+05***	Pu-238
Pu-239	***6E+03***	5E+04	***6E+03***	5E+04	Pu-239
Pu-240	***6E+03***	5E+04	***7E+03***	5E+04	Pu-240
Pu-241	***2E+05***	1E+06	***3E+05***	1E+06	Pu-241
Pu-242	***6E+03***	5E+04	***6E+03***	5E+04	Pu-242
Pu-244	***4E+02***	5E+04	***4E+02***	5E+04	Pu-244
Am-241	***7E+03***	5E+04	***9E+03***	5E+04	Am-241
Am-243	***9E+02***	4E+04	***9E+02***	4E+04	Am-243
Cm-243	1E+06	***8E+05***	5E+06	***8E+05***	Cm-243
Cm-244	***2E+06***	3E+06	***2E+06***	3E+06	Cm-244
Cm-245	***1E+03***	4E+04	***1E+03***	4E+04	Cm-245
Cm-246	***6E+03***	5E+04	***6E+03***	5E+04	Cm-246
Cm-247	***3E+02***	5E+04	***3E+02***	5E+04	Cm-247
Cm-248	***2E+03***	1E+04	***2E+03***	1E+04	Cm-248
Cf-249	***8E+02***	4E+04	***1E+03***	4E+04	Cf-249
Cf-250	***2E+06***	9E+06	***2E+06***	9E+06	Cf-250
Cf-251	***2E+03***	4E+04	***2E+03***	4E+04	Cf-251

a "NL" means No Limit - estimated permissible concentration is higher than the specific activity of the pure elemental radionuclide.

Eliminating the concentration limits for the homesteader scenario results in all radionuclide waste concentrations being based on the post-drilling scenario. The net result is an increase in 47 and 43 permissible radionuclide concentrations in the waste for the generic trench and tumulus designs, respectively. The changes in permissible concentrations range from none for Sm-151 (in the case of the generic trench) to an increase by three orders of magnitude for Al-26 (in the case of the generic trench and tumulus) and for Nb-94 (in the case of the generic trench). Shorter-lived radionuclides (i.e., radionuclides with half-lives shorter than about 30 y) are generally already limited by the post-drilling scenario and are not affected by elimination of the homesteader scenario. The medium- and longer-lived radionuclides are generally limited by the homesteader scenario because more restrictive exposure assumptions are used, and the radioactive decay differences due to the different times of exposure are minor. In general, the permissible waste concentrations for the homesteader and post-drilling scenarios are similar for the longer-lived radionuclides and radionuclides with longer-lived decay products. Therefore, the medium half-life radionuclides generally exhibit the most change in permissible waste concentrations when changing from the homesteader to the post-drilling scenario.

In summary, the permissible waste concentrations for intrusion for 47 and 43 of the 58 radionuclides for the generic trench and tumulus designs, respectively, are limited by the homesteader scenario. Disposal of the wastes below grade at a depth which precludes exposure via the homesteader scenario effectively eliminates this scenario as a realistic scenario. The permissible waste concentrations for intrusion scenarios would then be based solely on the post-drilling scenario and will be up to three orders of magnitude higher than the homesteader permissible waste concentrations. Disposal at a depth below grade that will preclude intrusion via the homesteader scenario appears to be a very effective method for increasing the permissible waste concentrations for intrusion. The sites most likely to benefit from this approach are generally arid sites which have sufficient vadose zone thickness for deeper disposal of wastes and for which the limits on permissible radionuclide concentrations in the waste are based on estimates derived using the intrusion scenario.

6.1.4.2 Time of Intrusion

Intrusion is assumed to be credible after loss of active institutional controls at 100 y following closure. The post-drilling scenario is assumed to occur at 100 y for sites with technology suitable for drilling water wells in hard rock. In regions where hard-rock drilling is uncommon (SRS), the time of post-drilling intrusion is assumed to occur at 300 y. Intrusion via the homesteader scenario is assumed to occur at 300 y for the generic trench and 500 y for the generic tumulus for all sites. The 300- and 500-y intrusion times are based on the assumed integrity of the trench and tumulus disposal facilities. The effect of the assumed time of intrusion is illustrated in Table 6-4 in which the permissible waste concentrations for the intrusion scenarios are presented for intrusion at 300 y for the generic trench and 500 y for the generic tumulus and 100 y for both.

Table 6-4. Maximum Permissible Waste Concentrations for Homesteader Intrusion at 100, 300, and 500 y (Part 1 of 2)

Nuclide	$C_{W\text{-}Intr}$ Homesteader ($\mu Ci/m^3$) Intrusion=100 y	$C_{W\text{-}Intr}$ Homesteader[a] ($\mu Ci/m^3$) Intrusion=300 y	$C_{W\text{-}Intr}$ Homesteader[a] ($\mu Ci/m^3$) Intrusion=500 y	Nuclide
H-3	1E+07	1E+12	9E+16	H-3
C-14	1E+04	1E+04	1E+04	C-14
Al-26	5E+01	5E+01	5E+01	Al-26
Si-32	2E+03	9E+03	4E+04	Si-32
Cl-36	2E+02	2E+02	2E+02	Cl-36
K-40	7E+02	7E+02	7E+02	K-40
Co-60	3E+07	NL	NL	Co-60
Ni-59	3E+06	3E+06	3E+06	Ni-59
Ni-63	2E+06	8E+06	3E+07	Ni-63
Se-79	2E+05	2E+05	2E+05	Se-79
Sr-90	1E+04	1E+06	1E+08	Sr-90
Zr-93	5E+06	5E+06	5E+06	Zr-93
Nb-93m	8E+08	5E+12	2E+16	Nb-93m
Nb-94	9E+01	9E+01	1E+02	Nb-94
Tc-99	2E+04	2E+04	2E+04	Tc-99
Pd-107	6E+06	6E+06	6E+06	Pd-107
Ag-108m	2E+02	5E+02	1E+03	Ag-108m
Cd-113m	2E+05	4E+09	7E+13	Cd-113m
Sn-121m	1E+06	2E+07	2E+08	Sn-121m
Sn-126	8E+01	8E+01	8E+01	Sn-126
I-129	2E+03	2E+03	2E+03	I-129
Cs-135	2E+05	2E+05	2E+05	Cs-135
Cs-137	3E+03	3E+05	3E+07	Cs-137
Ba-133	5E+08	3E+14	NL	Ba-133
Sm-151	4E+07	2E+08	8E+08	Sm-151
Eu-152	2E+04	6E+08	2E+13	Eu-152
Eu-154	4E+05	4E+12	NL	Eu-154
Pb-210	1E+04	7E+06	4E+09	Pb-210
Ra-226	8E+01	8E+01	9E+01	Ra-226
Ra-228	9E+06	NL	NL	Ra-228
Th-229	5E+02	5E+02	5E+02	Th-229
Th-230	2E+03	8E+01 [b]	8E+01 [b]	Th-230
Th-232	6E+01	6E+01	6E+01	Th-232
Pa-231	2E+02	2E+02	2E+02	Pa-231

Table 6-4. Maximum Permissible Waste Concentrations for Homesteader Intrusion at 100, 300, and 500 y (Part 2 of 2)

Nuclide	$C_{W\text{-}Intr}$ Homesteader ($\mu Ci/m^3$) Intrusion=100 y	$C_{W\text{-}Intr}$ Homesteader[a] ($\mu Ci/m^3$) Intrusion=300 y	$C_{W\text{-}Intr}$ Homesteader[a] ($\mu Ci/m^3$) Intrusion=500 y	Nuclide
U-232	2E+02	2E+03	1E+04	U-232
U-233	1E+04	7E+02 [c]	7E+02 [c]	U-233
U-234	2E+04	1E+03 [c]	1E+03 [c]	U-234
U-235	1E+03	6E+02 [c]	6E+02 [c]	U-235
U-236	2E+04	2E+04	2E+04	U-236
U-238	5E+03	5E+03	5E+03	U-238
Np-237	4E+02	4E+02	4E+02	Np-237
Pu-238	2E+04	7E+04	4E+05	Pu-238
Pu-239	6E+03	6E+03	6E+03	Pu-239
Pu-240	6E+03	6E+03	7E+03	Pu-240
Pu-241	2E+05	2E+05	3E+05	Pu-241
Pu-242	6E+03	6E+03	6E+03	Pu-242
Pu-244	4E+02	4E+02	4E+02	Pu-244
Am-241	5E+03	7E+03	9E+03	Am-241
Am-243	9E+02	9E+02	9E+02	Am-243
Cm-243	2E+04	1E+06	5E+06	Cm-243
Cm-244	4E+05	2E+06	2E+06	Cm-244
Cm-245	1E+03	1E+03	1E+03	Cm-245
Cm-246	6E+03	6E+03	6E+03	Cm-246
Cm-247	5E+02	3E+02	3E+02	Cm-247
Cm-248	2E+03	2E+03	2E+03	Cm-248
Cf-249	5E+02	8E+02	1E+03	Cf-249
Cf-250	1E+06	2E+06	2E+06	Cf-250
Cf-251	1E+03	2E+03	2E+03	Cf-251

a "NL" means No Limit - estimated permissible waste concentration is larger than the specific activity of the pure elemental radionuclide
b Concentration based on exposure at 9,000 y, the time of maximum dose
c Concentration based on exposure at 10,000 y, the time of maximum dose

Because the only difference in the scenarios is the time of intrusion, the short-lived radionuclides are most affected due to differences in their rates of decay. Additionally, because the 100-y homesteader intrusion would occur at the same time as the post-drilling scenario and because the homesteader scenario contains more restrictive exposure pathways, the 100-y homesteader scenario would provide slightly more restrictive permissible waste concentrations than the post-drilling scenario. The permissible waste concentrations for intrusion scenarios for longer-lived radionuclides and radionuclides with longer-lived decay products are generally not affected by reducing the times of intrusion from 300 and 500 y to 100 y. So for the PE, earlier intrusion would significantly affect only the shorter-lived radionuclides (i.e., those with half-lives less than about 100 y), and the permissible waste concentrations would be more restrictive by one to three orders of magnitude.

6.2 PARAMETER SENSITIVITY ANALYSIS

A parameter sensitivity analysis for the PE is presented in this section for the models used in the water and atmospheric pathway analysis and the intruder scenario analysis. These analyses show that the site-specific results are not sensitive to many generic and site-specific parameters, and those parameters that do not significantly impact the results are discussed and their dismissal justified in this chapter.

For those parameters which significantly affect the results of the analyses, the effects of their variation on the site-specific PE results are discussed in the individual site chapters. The parameter sensitivity analysis presented here separately addresses the water pathway, the atmospheric pathway, and the intrusion scenarios.

6.2.1 Water Pathway

The general equation used in the water pathway analysis is shown in Figure 6-1. Site-specific modifications of this equation at Oak Ridge and West Valley to include a shallow subsurface flow zone cause minor variations for pathways. The changes in results based on changes of individual parameters can be inferred from this equation. Each generic and site-specific parameter is discussed separately below. Because the retardation equation is contained within the decay equation, all of the parameters appear in the exponential of the decay equation.

6.2.1.1 Generic Parameters

The main objective of the PE is to provide information that can be used to compare the capabilities of various DOE facilities to serve as sites for the disposal of MLLW. Therefore, as long as waste forms and facility designs are consistent, the actual values are not as important. However, the PE provides a perspective on whether intrusion scenarios or transport scenarios (i.e., water or atmospheric transport) are likely to control waste concentrations. For this reason, reasonable values for the five generic parameter values (θ_G, K_d^G, ρ_G, f_m, and A) in the water pathway analysis are necessary for the results to be useful. These parameters are discussed in this section.

$$C_{W-Water} = 1000\, H_W \frac{1}{PDCF} \boxed{CRF_{Source}} \boxed{CRF_{Water}} \boxed{r_{Decay}}$$

$$CRF_{Source} = \frac{\theta_G + K_d^G \rho_G}{f_m}$$

$$CRF_{Water} = \frac{q_{gw}\, d_m\, a_2}{q_f\, A} + 1$$

$$r_{Decay} = \exp\left[\frac{\ln 2}{t_{1/2}}\left[t_{Detention} + \frac{l\,\theta_w}{q_f}\left(1 + \frac{K_d\,\rho_b}{\theta_w}\right) + \frac{100\,n}{q_{gw}}\left(1 + \frac{K_d\,\rho_b}{n}\right)\right]\right]$$

Figure 6-1. Major components and parameters used in the calculation of the permissible waste concentration for the water pathway. Parameters are explained in the Nomenclature listed at the front of this volume.

Grout Volumetric Moisture Content, θ_G

The grout volumetric moisture content, θ_G, appears only in the equation for CRF_{Source} (see Figure 6-1). Variations in θ_G values affect the results as the grout distribution coefficient, K_d^G, value becomes small. Under these conditions, the CRF_{Source} is proportional to changes in θ_G. A value of 0.3 is assumed in the analysis. The parameter has practical physical bounds ranging from about 0.1 to 0.3 (Glasser and Adkins, 1994). In regard to sensitivity of results, this parameter is of minor importance because, even when $K_d^G = 0$, the maximum variability in CRF_{Source} is a decrease by a factor of less than two.

Grout Distribution Coefficient, K_d^G

The grout distribution coefficient, K_d^G, appears only in the equation for CRF_{Source}. Changes in CRF_{Source} are proportional to changes in K_d^G for all but very low parameter values, which are influenced by the grout moisture content. The K_d^G values assumed for the different radionuclides are zero or multiples of ten to represent the potentially significant variability of this parameter. Values for this parameter have a wide degree of variability, and the values used in the PE were based on the conservative (i.e., smaller) values that were used in the Oak Ridge SWSA 6 performance assessment (ORNL, 1994). Values used in performance assessments at Savannah River (MMES, 1994) and Hanford (Kincaid et al., 1993) are sometimes two orders of magnitude larger than the values used in the PE, depending on the radionuclide. These higher values may include the effects of grout blends specifically designed to retain certain radionuclides. An order-of-magnitude change in the parameter will result in an order-of-magnitude change in the results.

Because the values for this parameter have a high degree of variability, it is an important parameter in regard to sensitivity. Additionally, because generic values were used, variations in this parameter produce the same results for the CRF_{Source} at all sites. Therefore, no other sensitivity analysis was performed.

Grout Bulk Density, ρ_G

The grout bulk density, ρ_G, appears only in the equation for CRF_{Source}. Changes in results are proportional to changes in ρ_G except for K_d^G values of 1 mL/g, which are also influenced by the grout moisture content, and K_d^G values of 0 mL/g, for which there is no effect. A value of 1.8 g/cm^3 is assumed in the PE analysis. The parameter has practical physical bounds ranging from about 1.1 to 2.5 g/cm^3 (Winter and Nilson, 1979). In regard to sensitivity of results, this parameter is of minor importance because the maximum variability in CRF_{Source} is an increase or decrease by a factor of less than two.

Mixing Fraction, f_m

The mixing fraction, f_m, appears only in the equation for CRF_{Source}. Because the parameter is in the denominator, changes in results are inversely proportional to changes in parameter values. Values of 0.33 and 0.67 are assumed in the analysis for the generic tumulus and trench facility, respectively. The parameter has practical physical bounds (i.e., the volume fraction of waste that can be emplaced in the facility) ranging from about 0.25 to 0.50 for the tumulus design and 0.4 to 0.75 for the trench design. In regard to sensitivity of results, this parameter is of minor importance because the maximum change in results is an increase or decrease by a factor of less than two.

Disposal Facility Plan Area, A, and Facility Width, a_2

These parameters appear only in the equation for CRF_{Water}. Assuming that the shape of the facility remains square, then changes in results are generally proportional to the inverse square root of the change in the area, A. Doubling the size of the disposal facility causes a reduction in the permissible waste concentration for the water pathway that is slightly smaller than the square root of the change in A. The exact effects of variations in the value for this parameter are site-specific because the modifying factors in the CRF_{Water} are site-specific.

The length of the disposal facility parallel to the flow of groundwater is the parameter implicit in the facility plan area of most importance. A longer length will lead to proportionally more leachate with respect to the groundwater flux. Increasing the length of the disposal facility parallel to groundwater flow will decrease the permissible waste concentrations. The practical consequence of this effect is that disposal facilities oriented with the long dimension perpendicular to groundwater flow will result in higher permissible waste concentrations than facilities oriented with the long dimension parallel to groundwater flow. Because the disposal facilities evaluated in the PE are hypothetical, the orientation problem is avoided by assuming a square-shaped plan area oriented parallel to the direction of groundwater flow.

Because the capacity of site-specific disposal facilities may need to be much different than the assumed facility, this parameter is potentially important in evaluating the permissible waste concentrations for a specific site. The effect of facility size is discussed in the site chapters.

6.2.1.2 Site-Specific Parameters

Many parameter values in the water pathway analysis are unique to each site. Some of these parameters have fairly narrow practical physical bounds, which limits their potential impact on the results. The discussions in this section illustrate the lack of sensitivity of the results to these parameters. Some other parameters have potentially more variability, although site characterization efforts can help narrow the uncertainty of these values. The trends in sensitivity are discussed in this section, and the effects on the results of variations in these parameter values are discussed in the site chapters.

Groundwater Darcy Velocity, q_{gw}

The groundwater Darcy velocity, q_{gw}, appears in the equation for CRF_{Water} and the equation for r_{Decay} (see Figure 6-1). Changes in CRF_{Water} are generally proportional to changes in q_{gw} (i.e., increasing the groundwater velocity increases dilution). Changes in r_{Decay} due to changes in q_{gw} are inversely proportional to changes in the exponential function of the decay equation (i.e., increasing the recharge rate decreases the vadose zone travel time). This factor is more important for shorter-lived radionuclides because r_{Decay} is related to arrival time and hence, decay. The exact effects of changes in this parameter are site-specific because the modifying factors in the CRF_{Water} and r_{Decay} are site-specific. This parameter is potentially important in regard to sensitivity. The impact of these variations on the permissible waste concentrations is discussed in the site chapters.

Mixing Depth, d_m

The mixing depth, d_m, appears in the equation for CRF_{Water}. Changes in results are generally proportional to changes in q_{gw} (i.e., increasing the mixing depth increases the permissible waste concentration due to increased dilution). Values for this parameter vary between 1 and 15 m for the 15 sites, with the lower values being limited by aquifer thickness and the upper values being based on the estimated plume depth at the 100-m performance boundary. For those sites where the mixing depth is constrained by the aquifer thickness, very little variability can occur and the results will be insensitive to these minor variations. The mixing depth for thick aquifers was estimated using generic estimates for transverse dispersivity (EPRI, 1985) and a one-dimensional flow, three-dimensional transport code (PAGAN [Chu et al., 1991]). The mixing depths based on plume thicknesses at 100 m provided by this code are not sensitive to changes in input parameter values. The primary reason for this insensitivity is that there is insufficient distance for larger changes in plume thickness to develop. In regard to sensitivity of results, this parameter is of minor importance.

Natural Recharge through Local Soils, i

The natural recharge through local soils, i, appears in the equations for CRF_{Water} and r_{Decay} (the water flowing through the facility, q_f, is initially controlled by the RCRA cover [trench] or the concrete vault [tumulus] properties; when all engineered barriers have failed, q_f is assumed to be equal to the natural recharge through local soils, i). Changes in CRF_{Water} are generally inversely proportional to changes in i (i.e., increasing the recharge decreases dilution by groundwater). Changes in r_{Decay} due to changes in i are inversely proportional to changes in the exponential function of the decay equation (i.e., increasing the recharge decreases the vadose zone travel time). This factor is more important for shorter-lived radionuclides because r_{Decay} is related to arrival time and hence, decay. The exact effects of changes to this parameter are site-specific because the modifying factors in the CRF_{Water} and r_{Decay} are site-specific. This parameter is potentially important in regard to sensitivity because it is difficult to measure; few sites considered in the PE were able to estimate values of this parameter with a high degree of confidence. The impact of the variations of i on the permissible waste concentrations is discussed in the site chapters.

Distance Between the Disposal Facility and the Saturated Zone, l

The distance between the disposal facility and the saturated zone, l, appears in the equation for r_{Decay} and affects the travel time in the vadose zone. Changes in results due to changes in l are proportional to changes in the exponential function of the decay equation, which are generally important only for shorter-lived radionuclides; increases in l result in increases in permissible waste concentrations. In regard to uncertainty of results, this parameter is of minor importance because the parameter values are site-specific and generally easily and accurately determined from site data. As a result, the associated uncertainty in the values for these parameters is generally very low.

Vadose Zone Volumetric Moisture Content, θ_w

This parameter appears in the equation for r_{Decay} and affects the travel time in the vadose zone. Because the K_d portion of the equation provides a much larger effect, values for volumetric moisture content are only important when the vadose zone distribution coefficient, K_d, is very low. Changes in the estimates of the permissible concentrations of radionuclides in the waste due to changes in θ_w are proportional to changes in the exponential function of the decay equation, which are generally important only for shorter-lived radionuclides. Increases in θ_w result in increases in vadose zone travel time and permissible waste concentrations.

While θ_w has a physical range from about 0.05 to greater than 0.50 (Guymon, 1994), site-specific ranges are much more constrained. In humid regions, a reasonable range for θ_w may be 0.15 to 0.50. For a short-lived radionuclide with a small K_d (e.g., H-3) at a site with a θ_w of 0.3, a recharge of 0.1 m/y, and a distance between the trench and saturated zone of 10 m, the maximum change in the estimates of the permissible concentrations of radionuclides in the waste is an increase or decrease by a factor of about 3. In arid regions, a reasonable range for θ_w may be 0.05 to 0.15. For a short-lived radionuclide with a small K_d (e.g., H-3) at a site with a θ_w of 0.10, a recharge of 0.05 m/y, and a distance between the trench and saturated zone of 30 m, the results increase or decrease by a factor of 6. Results for the tumulus facility and for longer-lived radionuclides will be less sensitive to variations in this parameter because the relative influence of the variations on r_{Decay} is less. In regard to sensitivity of results, this parameter is of minor importance because reasonable changes in its value do not significantly change the estimates of the permissible concentrations of radionuclides in the waste.

Saturated Zone Porosity, n

The saturated zone porosity, n, appears in the equation for r_{Decay} and affects the travel time in the saturated zone. Because the K_d portion of the equation provides a much larger effect, the porosity value is only important when the saturated zone travel time is large with respect to the vadose zone travel time (a condition often found in humid sites) and when the saturated zone distribution coefficient, K_d, is very low. Changes in the estimates of the permissible concentrations of radionuclides in the waste due to changes in n are proportional to the exponential function of the decay equation, which are generally important only for shorter-lived

radionuclides, with increases in n resulting in increases in saturated-zone travel-time and the permissible concentrations of radionuclides in the waste.

While n has a physical range across all geologic materials from about 0.01 to greater than 0.50 (Guymon, 1994), site-specific ranges are much more constrained because site-specific geology is generally known. Porosities of specific geologic materials can generally be measured to within a few percent (Department of the Army, 1980). Therefore, variations in this parameter will be minor, and in regard to the sensitivity of results, this parameter is of minor importance.

Bulk Density in the Vadose and Saturated Zones, ρ_b

The bulk density of the materials in the vadose and saturated zones appears in the equation for r_{Decay} and affects the retarded travel time in the vadose and saturated zones, respectively. Changes in the estimates of the permissible concentrations of radionuclides in the waste due to changes in ρ_b modified by the detention time in the disposal facility are proportional to changes in the exponential function of the decay equation, except for low K_d values which are influenced by the moisture content or porosity.

The values used in the analyses are site-specific but the bulk density has practical physical bounds ranging from about 1.3 to 2.5 g/cm^3 (Guymon, 1994). Bulk densities of specific geologic materials are generally known within a few percent (Department of the Army, 1980). Because this parameter is in the exponential function of the decay term, the maximum change in the estimates of the permissible concentrations of radionuclides in the waste due to doubling or halving ρ_b is an increase or decrease by a factor of about seven. Therefore, variations in this parameter will be minor, and in regard to sensitivity of results, this parameter is of minor importance.

Distribution Coefficient in the Vadose and Saturated Zones, K_d

The distribution coefficient in the vadose and saturated zones, K_d, appears in the equation for r_{Decay} and is used to calculate the retarded travel time for each radionuclide. Changes in the estimates of the permissible concentrations of radionuclides in the waste due to changes in K_d are inversely proportional to changes in the exponential function of the decay equation, and this factor is more important for shorter-lived radionuclides because r_{Decay} is related to arrival time. Values for these parameters are site-specific and depend on a wide range of variables including the element under consideration, solution chemistry (e.g., pH, Eh, and ionic strength), the presence of competing compounds, the soil mineralogy, and the presence of organic compounds. The studies needed to generate K_d values are generally not good indicators for field conditions. Therefore, most sites do not have site-specific values and default literature values must be used. There is wide variability in K_d values even for site-specific soils and radionuclides; major increases in the values for this parameter result in estimated arrival times at the performance boundary in excess of 10,000 y. As a result, short-lived radionuclides will be particularly impacted by variations in K_d. The exact effects of changes to this parameter are site-specific because the modifying factors in the r_{Decay} are site-specific. This parameter is potentially important in regard to sensitivity. The impact of these variations on the permissible waste concentrations is discussed in the site chapters.

6.2.1.3 Summary of Parameter Sensitivity Analysis for the Water Pathway

The sensitivity of results to variations in values for thirteen generic and site-specific parameters has been discussed for the water pathway, and the water pathway estimates of the permissible concentrations of radionuclides in the waste have been shown to be relatively insensitive to variations in all but five parameters (grout distribution coefficient, K_d^G; natural recharge, i; groundwater Darcy velocity; q_{gw}; plan area of the disposal facility, A; and soil distribution coefficients, K_d). The other parameters change the estimates of the permissible concentrations of radionuclides in the waste by less than an order of magnitude when varied to their practical maximums and minimums. The trends in results based on variations in the five influential parameters have been identified and discussed. Because generic values were used, variations in the grout distribution coefficient will affect all sites in the same manner. For this reason, this parameter is not discussed further. The site-specific effects of the variations of the remaining four influential parameters is presented and discussed in each site chapter in Volume 3 of this report.

6.2.2 Atmospheric Pathway

The general equation used in the atmospheric pathway analysis is shown in Figure 6-2. The trend in changes in results based on changes of individual parameters can be inferred from this equation. Each generic and site-specific parameter is discussed separately below.

6.2.2.1 Generic Parameters

The values for many parameters related to the atmospheric pathway analysis are generic to the PE method and the same at all sites. These parameters are discussed in this section.

Soil Thickness Above the Disposal Facility, x

The soil thickness above the disposal facility, x, appears only in the equation for CRF_{Diff} (see Figure 6-2). Changes in the estimates of the permissible concentrations of radionuclides in the waste are proportional to changes in parameter values with increasing thickness resulting in increasing permissible concentrations. A value of 1 m is arbitrarily assumed in the PE analysis. In regard to uncertainty of the estimates of the permissible concentrations of radionuclides in the waste, this parameter is of minor importance because one meter is likely a lower bound value for the cover thickness. However, disposal at deeper depths would lower the diffusion rate and result in higher permissible waste concentrations.

Ratio of Water Density in Air to Liquid and Ratio of CO_2 Concentration in Air to Dissolved CO_2 Concentration in Water, r

These ratios appear in the denominator of the equation for CRF_{Diff}, and changes in results are inversely proportional to changes in these values. In regard to sensitivity of the estimates of the permissible concentrations of radionuclides in the waste, these parameters are of minor importance because the values come from standard handbooks and are generally very well known; therefore, very little change in permissible waste concentrations will result from small possible changes in these parameters.

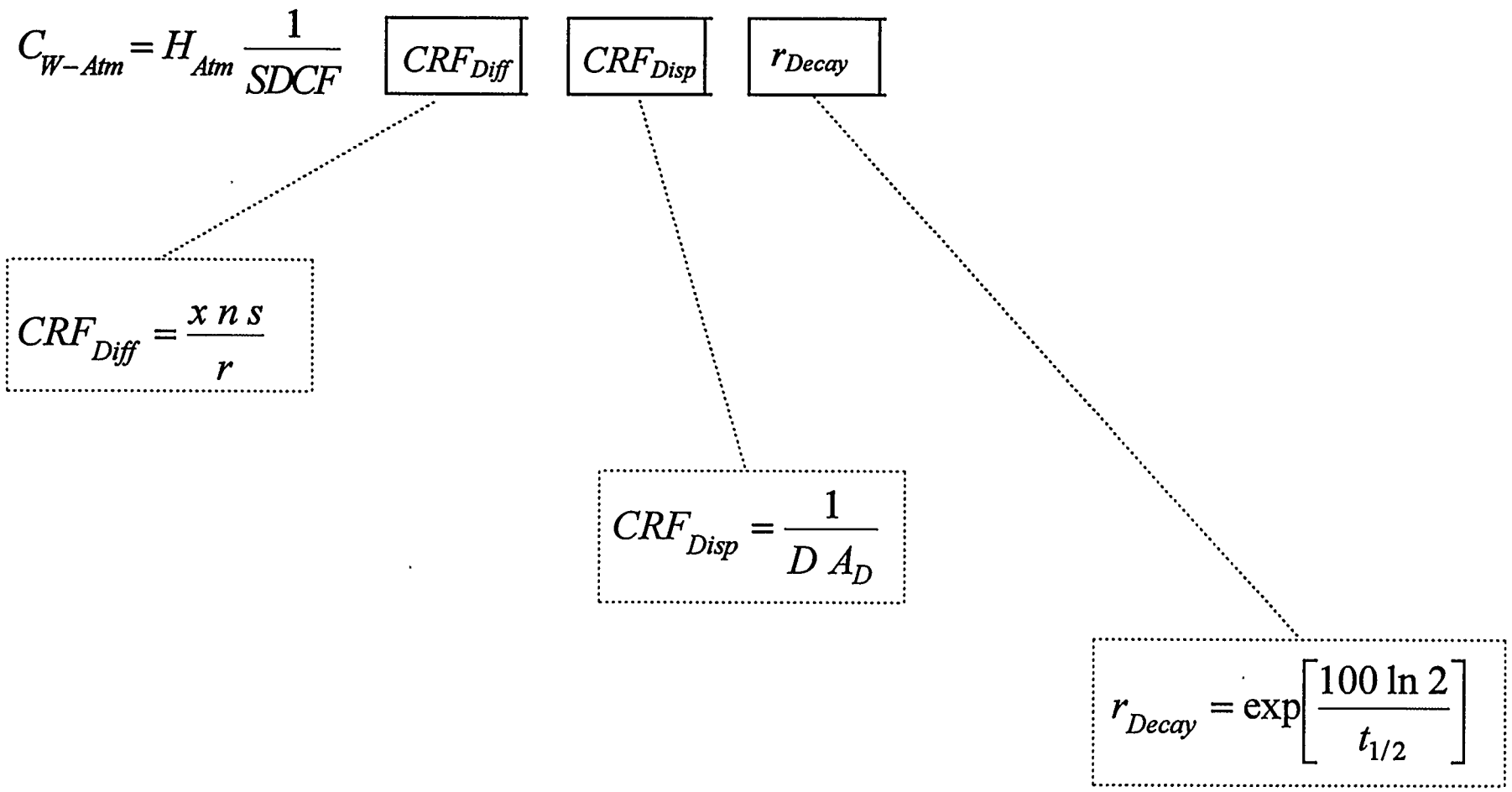


Figure 6-2. Major components and parameters used in the calculation of the permissible waste concentration for the atmospheric pathway. Parameters are explained in the Nomenclature listed at the front of this volume.

Tritium and Carbon Diffusion Coefficient in Air, D

These diffusion coefficients appear in the denominator of the equation for CRF_{Disp}, and changes in the estimates of the permissible concentrations of radionuclides in the waste are inversely proportional to changes in these values. In regard to sensitivity of the estimates of the permissible concentrations of radionuclides in the waste, these parameters are of minor importance because the values come from standard handbooks and are generally very well known; therefore, very little change in permissible waste concentrations will result from small possible changes in these parameters.

6.2.2.2 Site-Specific Parameters

Three parameter values in the atmospheric pathway analysis are unique to each site. These parameters have fairly narrow practical physical bounds, which limit the potential impact to the sensitivity of the results. The discussions in this section illustrates the lack of sensitivity of the results to these parameters.

Vadose Zone Porosity and Percent Saturation, n and s

These parameters appear in the equation for CRF_{Diff}, and variations in these parameters result in linear variations in the estimates of the permissible concentrations of radionuclides in the waste. The product of the vadose zone porosity and percent saturation equals the vadose zone volumetric moisture content, θ_w, and as discussed in Section 6.2.1, the moisture content has a physical range from about 0.05 to greater than 0.50, while site-specific ranges are much more constrained. In humid regions, a reasonable range for θ_w may be 0.15 to 0.60. In arid regions, a reasonable range for θ_w may be 0.05 to 0.15. In regard to sensitivity of the estimates of the permissible concentrations of radionuclides in the waste, this parameter is of minor importance because, for assumed moisture contents at the mid-point of these ranges, the variation in results when the parameter is varied to its extreme is a factor of about three.

Atmospheric Dispersion Term, A_D

This parameter appears in the denominator of the equation for CRF_{Disp}, and changes in the estimates of the permissible concentrations of radionuclides in the waste are inversely proportional to changes in these values. The CRF_{Disp} values provide minor concentration reductions relative to CRF_{Diff}. In addition, A_D varies over a fairly narrow range across all sites (minimum is 0.17 $(\mu Ci/m^3)/(\mu Ci/m^2\text{-s})$ at Argonne and maximum is 0.75 $(\mu Ci/m^3)/(\mu Ci/m^2\text{-s})$ at Oak Ridge). In regard to the sensitivity of the estimates of the permissible concentrations of radionuclides in the waste, this parameter is of minor importance because the 100-m performance boundary is so close to the disposal facility that significant dispersion effects are not developed, even with the very different wind speeds, directions, and stability classes represented by the 15 sites.

6.2.2.3 Summary of the Atmospheric Pathway

The sensitivity of six generic and site-specific parameters has been discussed for the atmospheric pathway, and the atmospheric pathway estimates of the permissible concentrations of

radionuclides in the waste have been shown to be relatively insensitive to variations in all parameters. None of the changes in site-specific parameters changed the estimates of the permissible concentrations of radionuclides in the waste by an order of magnitude even when varied to their practical maximums and minimums. Therefore, it can be concluded that the estimates of the permissible concentrations of radionuclides in the waste for the atmospheric pathway analysis are relatively insensitive to variations in the parameters used in this analysis. However, disposal at greater depths (increased value of x) results in lower diffusion rates and higher permissible concentrations of radionuclides in waste.

6.2.3 Intrusion Scenarios

The generic intruder scenario dose conversion factors (SDCFs) include consideration of doses from several exposure pathways. For the homesteader scenario, the exposure pathways are vegetable ingestion, soil ingestion, external exposure (in the garden and in the home), and inhalation of airborne soil particles (in the garden and in the home). The post-drilling scenario includes doses contributed from vegetable ingestion, soil ingestion, external exposure while working in the garden, and inhalation of airborne soil particles while in the garden. The parameters used to estimate doses from these exposure pathways are presented in Table 6-5. Also listed in Table 6-5 are the radionuclides that are controlled by each exposure pathway in the generic intruder scenario. In some cases, more than one pathway significantly contributes to the total dose received as a result of exposures to a single radionuclide.

The sensitivity of each SDCF (and hence, each radionuclide-specific waste concentration limit) to changes in values of specific input parameters depends on which exposure pathway is limiting. Radionuclides whose concentrations are limited by the ingestion pathway will be more sensitive to changes in the vegetable and soil intake rates and the mixing fraction of exhumed waste to clean soil; radionuclides whose concentrations are limited by the external exposure pathway will be more sensitive to the fraction of time exposed and the shielding factor provided by the home; those whose concentrations are limited by the inhalation pathway will be more sensitive to the inhalation rate and mass loading of soil particles in the air.

Summarized in the Table 6-6 are the effects of changes in the values for each for these input parameters on the magnitudes of the dose conversion factors for the dominant pathway of exposure for each of the eight indicator radionuclides. The analyses are based on evaluations of the generic trench homesteader intrusion scenario, with one parameter being modified at a time to evaluate the impacts on the SCDFs. The assumed changes in the parameters were selected to represent reasonable upper bounds based on physical limitations or other constraints.

As will be noted, the magnitudes of many of the SCDFs are directly related to the values of the key input parameters. This relationship is especially true for the short- to medium-lived radionuclides (i.e., H-3, C-14, and Sr-90) whose concentrations are limited by the ingestion pathway. While it also holds for a long-lived radionuclide (such as Tc-99), in the case of an extremely long-lived radionuclide (such as U-238) the relationship is less direct. A direct relationship also holds between the value of the SCDF and the exposure time for a radionuclide (such as Cs-137) whose concentration is limited by the external exposure pathway.

Table 6-5. Exposure Pathways, Pathway-Specific Parameters, and Pathway-Dominant Radionuclides

Exposure Pathway	Parameters	Nuclides Dominated by Specific Pathway
Vegetable ingestion	Intake rate for ingestion of vegetables (IR_v), density of soil, mixing fraction of exhumed waste to clean soil (f_m)	H-3, C-14, Si-32, Cl-36, Ni-59, Ni-63, Se-79, Sr-90, Zr-93, Nb-93m, Tc-99, Pd-107, Cd-133m, Sn-121m, I-129, Cs-135, Sm-151, Pb-210, U-232, U-236, U-238, Np-237 (22 nuclides)
Soil ingestion	Intake rate for ingestion of soil (IR_s), density of soil, mixing fraction of exhumed waste to clean soil (f_m)	Pa-231, U-236, Pu-238, Pu-239, Pu-240, Pu-241, Pu-242, Am-241, Cm-244, Cm-245, Cm-246, Cm-248, Cf-249, Cf-250 (14 nuclides)
External exposure (in the garden)	Mixing fraction of exhumed waste to clean soil (f_m), fraction of time exposed (f_{et})	Al-26, K-40, Co-60, Nb-94, Ag-108m, Sn-126, Cs-137, Ba-133, Eu-152, Eu-154, Ra-226, Ra-228, Th-229, Th-230, Th-232, Pa-231, U-232, U-233, U-234, U-235, U-238, Np-237, Pu-238, Pu-244, Am-243, Cm-243, Cm-247, Cf-249, Cf-251 (29 nuclides)
External exposure (in the home)	Mixing fraction of exhumed waste to clean soil (f_m), fraction of time exposed (f_{et}), shielding factor of home (f_s)	
Inhalation of airborne soil particles (in the garden)	Average inhalation rate (IR_a), mass loading of soil in air (L_a), fraction of time exposed (f_{et}), density of soil, mixing fraction of exhumed waste to clean soil (f_m)	Pa-231, U-236, Pu-238, Pu-239, Pu-240, Pu-241, Pu-242, Am-241, Cm-244, Cm-245, Cm-246, Cm-248, Cf-249, Cf-250 (14 nuclides)
Inhalation of airborne soil particles (in the home)	Average inhalation rate (IR_a), mass loading of soil in air (L_a), fraction of time exposed (f_{et}), density of soil	

Note: In some cases, doses from two or three pathways contribute significantly to the total dose for one radionuclide. Also, for Pu-238, soil ingestion and inhalation dominate before 3,000 y and external exposure dominates beyond 3,000 y; Cf-249 is dominated by external exposure prior to 3,000 y and by soil ingestion and inhalation after 3,000 y.

Table 6-6. Sensitivity of PE Scenario Dose Conversion Factors (SDCFs) for the Generic Trench Homesteader Intruder Scenario

	PE Results		Sensitivity to Variation in Parameters						
	Generic Intruder SDCF	Dominant Pathway	Double IR_v and IR_s	Increase IR_a (heavy exercise)	Increase f_s (no shielding from home)	Increase L_a 10-fold	Increase f_m (no mixing with clean soil)	Increase f_{et-g}	Increase f_{et-h}
Parameters:									
IR_v (kg/y)	90		**180**						
IR_s (kg/y)	0.037		**0.074**						
IR_a (m^3/y)	8000			**26000**					
f_s	0.7				**1**				
L_{a-g} (kg/m^3)	1E-7					**1E-6**			
L_{a-h} (kg/m^3)	1E-8					**1E-7**			
f_{m-ag}	0.2						**1.0**		
f_{et-g}	0.01							**0.25**	
f_{et-h}	0.5								**1**
H-3	4.0E-06	veg	7.9E-06	4.0E-06	4.0E-06	4.0E-06	2.0E-05	4.0E-06	4.0E-06
C-14	1.5E-05	veg	3.0E-05	1.5E-05	1.5E-05	1.5E-05	7.5E-05	1.5E-05	1.5E-05
Sr-90	2.0E-04	veg	4.0E-04	2.0E-04	2.0E-04	2.0E-04	9.9E-04	2.0E-04	2.0E-04
Tc-99	1.2E-05	veg	2.4E-05	1.2E-05	1.2E-05	1.2E-05	6.1E-05	1.2E-05	1.2E-05
Cs-137	7.6E-04	external	7.7E-04	7.6E-04	1.1E-03	7.6E-04	8.1E-04	8.5E-04	1.5E-03
U-238	4.2E-05	veg	5.0E-05	5.0E-05	5.5E-05	7.5E-05	7.4E-05	4.9E-05	7.7E-05
Pu-239	3.2E-05	soil/inhal	5.2E-05	6.1E-05	3.2E-05	1.5E-04	1.1E-04	4.4E-05	4.5E-05
Am-241	2.9E-05	soil/inhal	4.4E-05	4.7E-05	3.2E-05	1.0E-04	9.1E-05	3.7E-05	4.3E-05

IR_v = vegetable intake rate; IR_s = soil intake rate; IR_a = adult inhalation rate; f_s = shielding factor; L_{a-g} = mass loading in the garden; L_{a-h}= mass loading in the home; f_{m-ag} = mixing fraction of exhumed waste to soil for the agriculture intrusion scenario; f_{et-g} = fraction of time exposed to external radiation in the garden; f_{et-h}= fraction of time exposed to external radiation in the home.

7. SUMMARY OF RESULTS AND CONCLUSIONS

Based on technical considerations, estimates of the permissible waste concentrations for 58 radionuclides that can be placed in two types of generic MLLW disposal facilities at each of 14 DOE sites are presented in Volume 3 of this report. These estimates were based on the PE methodology presented in this volume (Volume 2). A 15th site (WVDP) was analyzed for only 18 radionuclides because these were the only ones expected to be in the MLLW at that site. The estimated permissible waste concentrations that would be acceptable for disposal at each site are presented in the chapters in Volume 3 that cover each site. These chapters in Volume 3 contain detailed descriptions of the conceptual models, assumptions, and input parameters used for the calculations for the water pathway, atmospheric pathway, and intruder scenarios. A discussion of the conceptual model assumptions and parameter sensitivity analyses of the estimates derived through the PEs is presented in Chapter 6 of this volume. Summarized in this chapter are the results of the analyses performed for all 15 sites. This summary includes a discussion of these estimates in context of the uncertainty and sensitivity associated with the analyses and with respect to site-specific performance assessments. It also includes a listing of the conclusions that were reached on the basis of this discussion.

The PE is focused on the disposal of radionuclides in the physical and chemical forms expected to be present in treated and stabilized DOE MLLW. Many important issues related to MLLW disposal have not been considered in this analysis, including distributive equity, MLLW treatment performance and costs, transportation risks and costs, and disposal performance of the RCRA constituents that will also be present in MLLW. These issues will be addressed in later phases of the planning process for MLLW disposal.

The performance evaluation represents a first-order scoping study only. The results are limited in applicability, having been developed for the sole purpose of comparing the various DOE sites on the basis of estimated limits on radionuclide concentrations in wastes that can be placed in hypothetical disposal facilities. The conduct of site-specific performance assessments prior to construction or operation of any DOE MLLW disposal facility is required by DOE Order 5820.2A. Since such assessments frequently account for additional concentration attenuation mechanisms, they generally result in estimated permissible waste concentrations that are less restrictive than those presented in this report.

7.1 INDICATOR RADIONUCLIDES

As indicated above, the estimated concentration limits for a given set of radionuclides in the DOE MLLW inventory are presented in Volume 3 for the two proposed disposal facility designs at each of the 15 sites. Each set of estimates is based on an analysis involving three exposure pathways. Although each of the radionuclides is unique, many of them are similar with respect to important characteristics, such as their half-lives, environmental mobility, and radiotoxicity.

To simplify the presentation of results and to facilitate the discussions in this chapter, each of the 58 radionuclides is aggregated into one or more of eight groups, with each of these groups being represented by an "indicator" radionuclide. These eight indicator radionuclides were selected because they (1) represent certain specific characteristics in terms of their half-lives, their environmental mobility, and their radiotoxicity, and (2) are some of the more important radionuclides that have been identified in DOE MLLW streams. After this general discussion using indicator radionuclides, a summary of the exposure pathways limiting the permissible waste concentrations based on the entire 58 radionuclides is presented. The eight indicator radionuclides and their associated characteristics are listed in Table 7-1.

Table 7-1. Characteristics of the Indicator Radionuclides

Radionuclide	Half-Life (years)		Mobility	Radiotoxicity
H-3	Short	12.3	High and Volatile	Low
C-14	Medium	5700	High and Volatile	Low
Sr-90	Short	29.1	High	Medium
Tc-99	Long	213,000	High	Low
Cs-137	Short	30.2	Medium	Medium
U-238	Long	4.47 billion	Medium	Medium
Pu-239	Long	24,100	Low	High
Am-241 (Np-237)[a]	Medium (Long)	433 (2.14 million)	Low (High)	High (High)

Half-life - Short: $t_{1/2} \leq 30$ y; Medium: $30 < t_{1/2} \leq 10{,}000$ y; Long $t_{1/2} > 10{,}000$ y
Mobility - High: $K_d \leq 5$ mL/g; Medium: $5 < K_d \leq 100$ mL/g; Low: $K_d > 100$ mL/g
Radiotoxicity - Low: PDCF ≤ 1 (rem/y)/(μCi/L); Medium: 1 < PDCF ≤ 100 (rem/y)/(μCi/L); High: PDCF > 100 (rem/y)/(μCi/L)
a Although Np-237 is a decay product of Am-241, it has significantly different properties. See text for discussion of this indicator radionuclide.

For the purposes of grouping of radionuclides, a short half-life is defined as less than approximately 30 y (where Cs-137 [$t_{1/2} = 30.2$ y] is classified as short-lived), medium as 30 to 10,000 y, and long as in excess of 10,000 y. While half-life is an easily quantifiable parameter, the terms short-, medium-, and long-lived are more arbitrary and should be viewed within the context of their travel time at a particular site.

Environmental mobility depends on many factors related to soil characteristics, the chemical form of the radionuclide in the environment, pH, Eh, and sorption potential. As modeled in the PE, environmental mobility is largely a function of the sorption coefficient (K_d) of the radionuclide in the soil, with lower K_d values resulting in higher mobility. Radionuclides are arranged into three groups in this chapter, with a high mobility radionuclide defined as having a K_d between 0 to 5 mL/g; medium mobility as a K_d between 5 to 100 mL/g; and low mobility as a K_d over 100 mL/g. Again, the terms low, medium, and high mobility are arbitrary.

Radiotoxicity is represented by pathway dose conversion factors (PDCF) for the water pathway (see Table 5-6 of this volume). The PDCFs for the 58 radionuclides used in the PE analysis range over six orders-of-magnitude. For the purposes of grouping the radionuclides, a low radiotoxicity radionuclide is defined as having a PDCF less than 1 (rem/y)/(μCi/L); medium toxicity as 1 to 10^2 (rem/y)/(μCi/L); and high toxicity as greater than 10^2 (rem/y)/(μCi/L). As in the case of environmental mobility, the terms low, medium, and high are arbitrary.

Tritium (H-3) and C-14 are included as indicator radionuclides in part because they are the only radionuclides being analyzed in the PE for the atmospheric pathway. Other radionuclides, including I-129 and Cs-137, could become volatile under high temperature conditions but are not expected to be volatile under the disposal facility conditions anticipated to be present at the sites being evaluated.

The remaining six indicator radionuclides represent various decay rates, mobility, and radiotoxicity. Strontium-90 and Cs-137 are short-lived radionuclides important in the DOE MLLW inventory and have medium and low environmental mobility, respectively. Because of its high mobility and long half-life, Tc-99 has traditionally been a problem radionuclide for performance assessments with respect to the water pathway. Two long-lived actinides (U-238 and Pu-239) are frequently present in the DOE MLLW inventory. The radionuclide Am-241, which has a "medium" half-life (430 y) and "low" mobility, has a long-lived decay product (Np-237) with much higher mobility. As a result, this decay product dominates the doses at later exposure times (after approximately 5000 y). Consequently, permissible waste concentrations for the water pathway for Am-241 are based on doses from Am-241 for early arrival times at the performance boundary but from Np-237 for later arrival times. The tables in this chapter that summarize the permissible concentrations for Am-241 identify which radionuclide is contributing the majority of the permissible dose.

While all 58 radionuclides cannot be definitively categorized into a single one of the eight indicator radionuclide groupings, a generalized grouping is shown in Table 7-2. As a consequence of the impact of significant decay products (discussed above), some of the radionuclides appear in more than one indicator radionuclide grouping. Because transport values of radionuclides depend on localized conditions, a site-specific grouping may be different than that presented here.

The remainder of this chapter presents an overall summary and discussion of the results of the PE analyses. A summary of the estimated permissible waste concentrations for the three individual pathways—water, atmospheric, and intrusion—is presented in Sections 7.2 through 7.4, with a comparison of all pathways being presented in Section 7.5. The estimated limits for the individual radionuclides are discussed in Section 7.6, and the conclusions reached as a result of the associated analyses are summarized in Section 7.7.

Table 7-2. A General Grouping of the 58 Radionuclides Evaluated in the PE

Indicator Radionuclide	Nuclides with Similar Half-Life and Mobility Characteristics
H-3	Volatile
C-14	Volatile
Sr-90	Co-60, Cd-113m, Ba-133
Tc-99	(I-129)[a], (Np-237), (Am-241)[b], (Pu-241)[b]
Cs-137	(Nb-93m), Eu-152, Eu-154, (Pb-210), (Ra-228)
U-238	(Al-26), (Cl-36), (K-40), (Pd-107), U-233, U-234, U-235, (U-236), Pu-238[c]
Pu-239	(Ni-59), (Se-79), (Zr-93), (Nb-94), (Sn-126), (Cs-135), Th-230, Th-232, Pa-231, Pu-242, Pu-244, Cm-243[d], Cm-247, Cm-248
Am-241[e]	(Si-32), (Ni-63), (Ag-108m), (Sn-121m), (Sm-151), Ra-226, Th-229, U-232, Pu-238, Pu-240, Pu-241[f], Am-243, Cm-244[g], Cm-245, Cm-246, Cf-249, Cf-250[h], Cf-251

a Parentheses indicate that toxicity category is different than for indicator radionuclide.
b Am-241 and Pu-241 are based on their decay product Np-237.
c The characteristics of Pu-238 are based on its decay product U-234.
d The characteristics of Cm-243 are based on its decay product Pu-239.
e The listed radionuclides are similar to Am-241 only for early arrival times at the performance boundary. Because these radionuclides have medium half-lives and are generally of low mobility, later arrival times will result in significant radioactive decay. Later arrival times for Am-241 result in its mobile, long-lived, radiologically significant decay product, Np-237, contributing significantly to the dose.
f The characteristics of Pu-241 are based on its decay product Am-241.
g The characteristics of Cm-244 are based on its decay product Pu-240.
h The characteristics of Cf-250 are based on its decay product Cm-246.

7.2 RESULTS OF THE WATER PATHWAY ANALYSIS

The water pathway analysis is based on site-specific modification of a generic conceptual model for flow and transport of radionuclides and uses a framework that provides consistency of analysis for the 15 sites. The attenuation in the concentrations between the disposed waste and the performance boundary is represented by the source concentration reduction factor, CRF_{Source}, the environmental transport concentration reduction factor for the water pathway, CRF_{Water}, and the radioactive decay term, r_{Decay}. The CRF_{Source} represents the concentration attenuation between the disposed waste and the leachate exiting the disposal facility and is modeled as a process that incorporates desorption and infiltration. The CRF_{Water} represents the attenuation in concentrations of the radionuclides between the leachate emerging from the waste and that in the groundwater reaching the 100-m performance boundary. It is modeled as a dilution of leachate in the groundwater flowing beneath the disposal facility. The r_{Decay} represents the radioactive decay that occurs prior to arrival at the performance boundary. The PE methodology is described in chapter 5 of this volume, and details of the site-specific analyses are contained in Volume 3 of this report.

7.2.1 Natural Site Characteristics

The performance evaluation showed that the estimates of permissible radionuclide concentrations in the waste, based on the water pathway, were highly dependent on some of the natural characteristics of the site. These include the recharge, depth-to-groundwater, and

subsurface geology. Of these, the annual recharge is directly affected, and the depth-to-groundwater is affected to some extent, by the climate of the region in which the disposal facility is located. For this reason, the 15 sites were divided into arid and humid groups, with the former including LLNL, Hanford, NTS, INEL, RFETS, SNL, LANL, and Pantex and the latter including ANLE, PGDP, FEMP, PORTS, ORR, SRS, and WVDP. Table 7-3 and Figure 7-1 show several important characteristics of the 15 sites analyzed by the PE. Sites being compared in the tables and figures in this chapter are arranged so that sites in the western portions of the contiguous U.S. are shown on the left and sites in eastern portions of the U.S. are shown on the right.

For three of the sites, the subsurface geology was a special consideration in determining flow and transport through the vadose zone. For LANL, ORR, and WVDP, portions of the geologic media in the vadose zone were not considered for various reasons (see footnotes to Table 7-3). The permissible waste concentrations for the water pathway at these sites reflect the reduced total thickness of the vadose zone used in the PE analyses.

The natural recharge is used to estimate the amount of water passing through the disposal facility after the engineered barriers are assumed to be no longer functioning. The recharge is also used to estimate the volume of leachate moving through the vadose zone. Estimated recharge ranges from less than 0.01 m/y at Pantex to 0.40 m/y at SRS. No recharge is reported for NTS because the draft performance assessment for the LLW facility at that site indicates that no net recharge occurs. The natural recharge is determined by many factors including precipitation, evapotranspiration, surface topography and runoff, and the hydraulic conductivity of the subsurface geology. For example, ORR, which has the highest precipitation rate of all sites, has less than half the natural recharge of SRS due to the low hydraulic conductivity of the underlying formations and a hilly terrain that promotes surface and shallow subsurface runoff. As illustrated in Figure 7-1(a), with the exception of WVDP, which overlies a formation with very low hydraulic conductivity, most eastern sites have higher natural recharge than western sites.

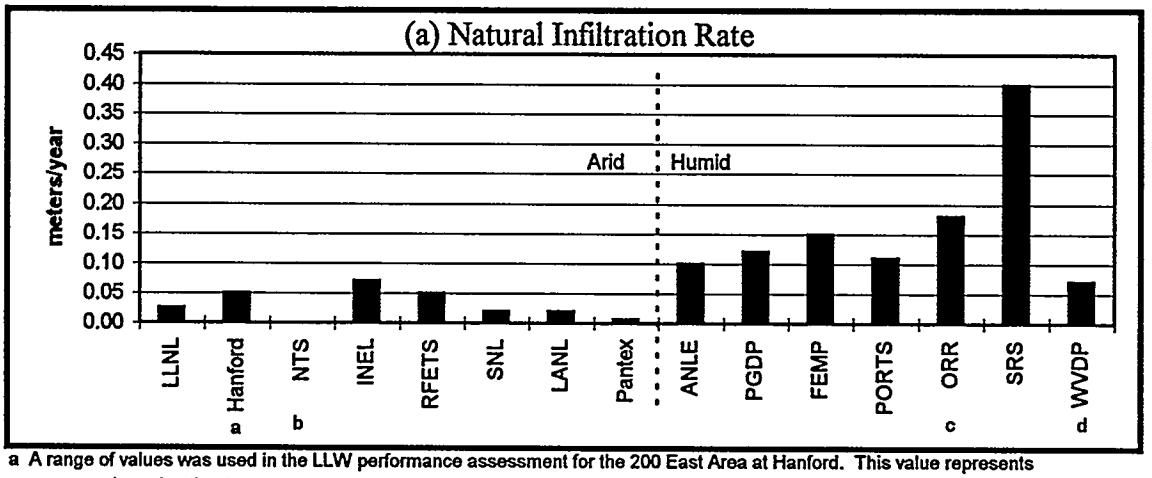
The depth-to-groundwater (Figure 7-1[b]) controls the time required for water and radionuclides to reach the groundwater. At the 15 sites, this depth, commonly referred to as the "vadose zone," ranges from 1.5 m at PGDP to 360 m at LANL. With the exception of RFETS, the western sites generally have much thicker vadose zones than the eastern sites. To assure conservatism in the analyses, the thickness of the vadose zones at LANL, ORR, and WVDP were assumed to be equal to the non-fractured portion only (see Table 7-3).

The groundwater Darcy velocity (Figure 7-1[c]) is related to the amount of leachate dilution provided by the groundwater and to the time required for water and the accompanying radionuclides to travel through the saturated zone to the 100-m performance boundary. The groundwater Darcy velocities for the 15 sites range from 0.31 m/y at WVDP to 56 m/y at INEL. These velocities are determined by the hydraulic gradient and the hydraulic conductivity at each site. Following this approach, the relatively flat hydraulic gradient at SNL and the low hydraulic conductivity at WVDP both result in low groundwater Darcy velocities.

Table 7-3. Characteristics of the 15 Sites

Parameter	Arid Sites								Humid Sites						
	LLNL	Hanford	NTS	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
Natural recharge (m/y)	0.025	0.05	- [a]	0.07	0.05	0.02	0.02	0.006	0.1	0.12	0.15	0.11	0.18 [b]	0.4	0.01 [c]
Depth to Groundwater (m)	160	72	240	50	13	150	360 [d]	120	38	1.5 [e]	6.7	5	2 [f]	17	1 [g]
Groundwater Darcy velocity (m/y)	9.9	22	1.4	56	0.59	0.5	23	6	6	25	28	7.7	2.9	8.1	0.31
Water travel time in vadose zone (y)	1100	130	65,000	120	31	530	830	2000	112	26 [h]	11	10	- [i]	8.5	- [i]
Water travel time in saturated zone to 100-m performance boundary (y)	2.5	1.4	70	0.18	17	60	1.3	4	0.5	1	1.1	4.5	1.2	3.7	670 [j]

a The draft LLW performance assessment indicates no net downward water migration. Travel time is therefore a conservative estimate provided in the draft performance assessment.
b Recharge through the disposal facility due to contributions from up-slope runoff is estimated to be 2.2 m/y. The majority of infiltratng water flows laterally in the shallow subsurface, so that only 0.18 m/y contributes to the leachate flux entering the groundwater system.
c Natural recharge is 0.07 m/y, with vertical deep recharge equal to 0.01 m/y and the remaining water flowing laterally.
d Vadose zone thickness used in the PE is 333 m, which is the result of not considering 27 m of fractured tuff in the transport analysis.
e Depth to groundwater is 1.5 m, but depth to the Regional Gravel Aquifer is 15.5 m, which is the result of adding 1.5 m of unsaturated material to 14.0 m of saturated media in which transport is predominantly vertical.
f Vadose zone thickness used in the PE is 0 m, which is the result of not considering 2 m of fractured saprolite in the transport analysis.
g Vadose zone thickness used in the PE is 0 m, which is the result of not considering 1 m of material in the transport analysis.
h Water travel time to the water table is 2.5 y for the tumulus, but travel time to the Regional Gravel Aquifer is 26 y.
i Data not available or estimated.
j Includes vertical (540 y) and horizontal (130 y) flow path.

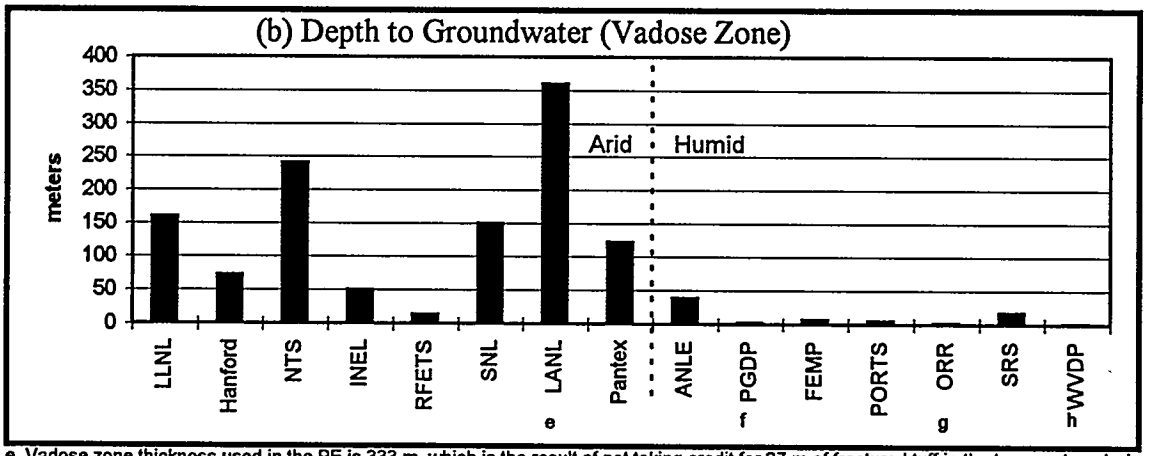


a A range of values was used in the LLW performance assessment for the 200 East Area at Hanford. This value represents an upper bound estimate.

b The draft performance assessment indicates a lack of net downward migration at NTS.

c Infiltration through the disposal facility due to contributions from up-slope runoff is estimated to be 2.2 m/y.

d Natural infiltration rate is 0.07 m/y, with vertical deep infiltration equal to 0.01 m/y and the remaining infiltration flowing laterally.



e Vadose zone thickness used in the PE is 333 m, which is the result of not taking credit for 27 m of fractured tuff in the transport analysis.

f Vadose zone thickness used in the PE is 15.5 m, which is the result of adding 1.5 m of unsaturated material to 14.0 m of saturated media in which transport is predominantly vertical.

g Vadose zone thickness used in the PE is 0 m, which is the result of not taking credit for 2 m of fractured saprolite in the transport analysis.

h Vadose zone thickness used in the PE is 0 m, which is the result of not taking credit for 1 m of fractured material in the transport analysis.

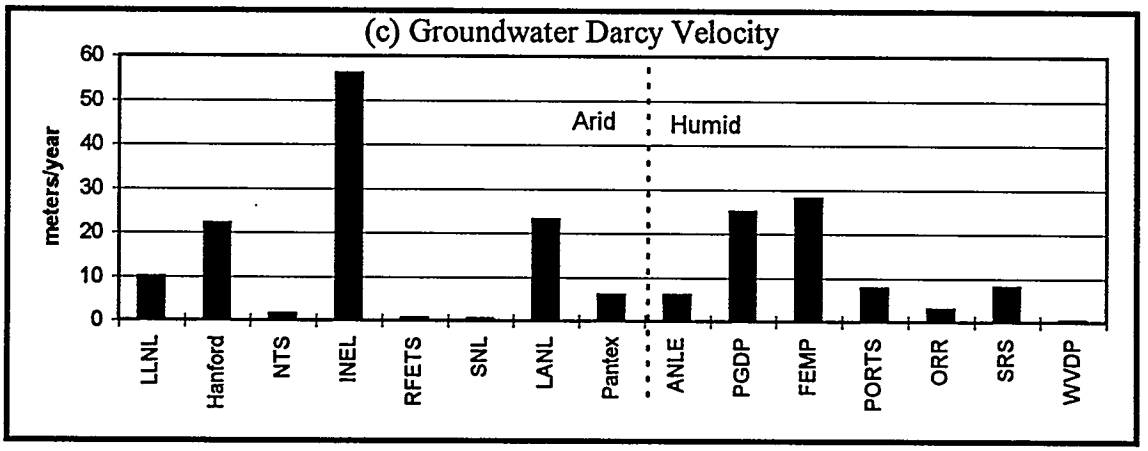


Figure 7-1. Selected characteristics of the 15 sites.

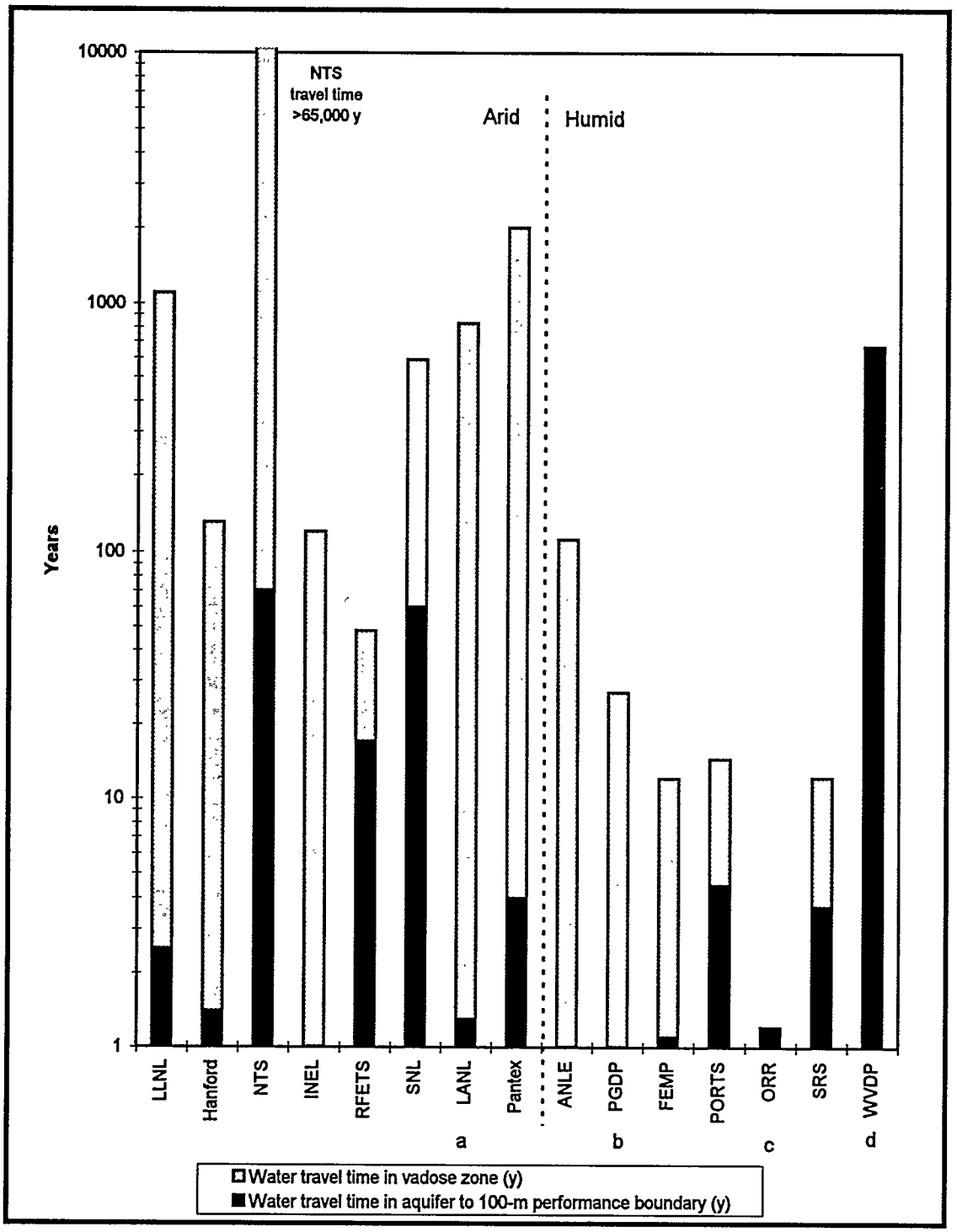
Water travel times in the vadose and saturated zones are presented in Table 7-3 and illustrated in Figure 7-2. For sites with thick vadose zones (Figure 7-1[b]) (i.e., LLNL, Hanford, NTS, INEL, SNL, LANL, Pantex, and ANLE), the estimated travel times through the vadose zone are 9 to 900 times larger than those for the saturated zone. The SNL ratio of 9 is due to the low hydraulic gradient in the regional aquifer; this low gradient results in low groundwater velocities. The remaining sites have much thinner vadose zones, and the travel times in the vadose and saturated zones are on the same order of magnitude. The long water travel time in the saturated zone at WVDP is due to the low hydraulic conductivity of one formation and the low hydraulic gradient in the aquifer.

These site characteristics are generally similar for the humid sites (high recharge and thin vadose zones) and the arid sites (low recharge and thick vadose zones), although there are some exceptions. For example, the vadose zone is relatively thin at RFETS although it is otherwise consistent with an arid site, and the recharge at WVDP is relatively low for a humid site. As may be noted in the discussion that follows, the permissible waste concentrations for most radionuclides for disposal in facilities at arid sites are limited by the intrusion scenario, while at humid sites the permissible concentrations of many of the radionuclides are limited by the water pathway. Because the intrusion scenarios are essentially identical for the 15 sites, sites where the waste concentrations are limited by the water pathway will have more restrictive limits than those where the waste concentrations are limited by intrusion. This condition implies that, based on these considerations, concentration limits for disposal of radionuclides in humid regions will generally be more restrictive than those in arid regions.

The presence of certain natural characteristics at disposal sites can improve their performance relative to retarding the migration of radionuclides. For example, if a disposal facility is underlain with a sand containing clay having a large sorption capacity, the movement of radionuclides may be significantly retarded relative to a clean sand even though water travel times may be short. For this reason, it is important to consider site-specific factors in conducting performance analyses. Such factors were considered to the extent possible in conducting the PEs while, at the same time, care was taken to maintain a consistent framework throughout the analyses.

7.2.2 Results for the Water Pathway

A summary of the permissible concentrations for each of the eight indicator radionuclides, based on analyses for the water pathway, for each of the 15 sites is shown in Table 7-4. The limits for the generic trench are shown in part (a) and for the generic tumulus in part (b). Each indicator radionuclide will be discussed individually later in this section in the context of the site- and radionuclide-specific characteristics, but some general conclusions related to Table 7-4 are presented here.



a Vadose zone thickness used in the PE is 333 m, which is the result of not taking credit for 27 m of fractured tuff in the transport analysis.

b Vadose zone thickness used in the PE is 15.5 m, which is the result of adding 1.5 m of unsaturated material to 14.0 m of saturated media in which transport is predominantly vertical.

c Vadose zone thickness used in the PE is 0 m, which is the result of not taking credit for 2 m of fractured saprolite in the transport analysis.

d Vadose zone thickness used in the PE is 0 m, which is the result of not taking credit for 1 m of fractured material in the transport analysis.

Figure 7-2. Subsurface water travel times (y) in the vadose and saturated zones.

Table 7-4. Permissible Waste Concentrations for the Water Pathway ($\mu Ci/m^3$)

(a) Generic Trench

Nuclide	Arid Sites								Humid Sites							Nuclide
	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP	
H-3	NL [b]	3E+07	-	1E+08	7E+04	NL	NL	NL	5E+06	5E+05	6E+04	4E+03	2E+03	7E+04	4E+11	H-3
C-14	1E+07 [c]	3E+03	-	1E+04	3E+02	2E+06 [c]	4E+08 [c]	NL [c]	5E+02	3E+03	9E+02	3E+02	1E+02	4E+02	4E+02	C-14
Sr-90	NL [c]	NL	-	NL	NL	NL [c]	NL [c]	NL [c]	NL	NL	9E+10	3E+05	NL	6E+11	NL	Sr-90
Tc-99	1E+03	5E+02	-	2E+03	4E+01	1E+02	2E+03	3E+03	5E+01	4E+02	1E+02	5E+01	2E+01	6E+01	6E+01	Tc-99
Cs-137	NL [c]	NL	-	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL	NL [c]	NL	NL [c]	Cs-137
U-238	4E+02 [c]	2E+02	-	1E+03 [c]	2E+01 [c]	4E+01 [c]	1E+03 [c]	1E+03 [c]	2E+01 [c]	2E+02	6E+01	2E+01	1E+01	3E+01	2E+01 [c]	U-238
Pu-239	NL [c]	4E+03 [c]	-	NL [c]	3E+02 [c]	NL [c]	NL [c]	NL [c]	1E+04 [c]	8E+01 [c]	1E+01 [c]	3E+00 [c]	9E-01	3E+00	NL [c]	Pu-239
Am-241	1E+05 [c,d]	7E+04 [c,d]	-	3E+05 [c,d]	6E+03 [d]	1E+04 [c,d]	3E+05 [c,d]	4E+05 [c,d]	7E+03 [d]	6E+04 [d]	2E+04 [d]	7E+03 [d]	2E+01	8E+03 [d]	8E+03 [d]	Am-241

(b) Generic Tumulus

Nuclide	Arid Sites								Humid Sites							Nuclide
	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP	
H-3	NL	1E+14	-	3E+14	2E+12	NL	NL	NL	1E+13	1E+12	7E+10	3E+10	3E+06	2E+10	NL	H-3
C-14	4E+07 [c]	7E+03	-	3E+04	8E+02	6E+06 [c]	8E+08 [c]	NL [c]	1E+03	7E+03	2E+03	7E+02	3E+02	8E+02	1E+03	C-14
Sr-90	NL [c]	NL	-	NL	NL	NL [c]	NL [c]	NL [c]	NL [c]	NL	NL	NL	NL	NL	NL [c]	Sr-90
Tc-99	2E+03	1E+03	-	5E+03	9E+01	2E+02	5E+03	5E+03	1E+02	9E+02	3E+02	1E+02	5E+01	1E+02	1E+02	Tc-99
Cs-137	NL [c]	NL	-	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	NL [c]	Cs-137
U-238	9E+02 [c]	5E+02	-	2E+03 [c]	4E+01 [c]	9E+01 [c]	2E+03 [c]	2E+03 [c]	5E+01 [c]	4E+02	1E+02	5E+01	2E+01	5E+01	5E+01 [c]	U-238
Pu-239	NL [c]	2E+04 [c]	-	NL [c]	3E+04 [c]	NL [c]	NL [c]	NL [c]	3E+05 [c]	1E+03 [c]	4E+01 [c]	2E+01 [c]	2E+00	6E+00 [c]	NL [c]	Pu-239
Am-241	3E+05 [c,d]	1E+05 [c,d]	-	6E+05 [c,d]	1E+04 [d]	3E+04 [c,d]	7E+05 [c,d]	8E+05 [c,d]	1E+04 [d]	1E+05 [d]	4E+04 [d]	1E+04 [d]	5E+01	2E+04 [d]	2E+04 [c,d]	Am-241

a Water pathway was not evaluated for this site.
b No Limit - estimated permissible waste concentration is greater than the specific activity of the pure elemental radionuclide.
c Radionuclide concentration is based on an arrival time beyond 10,000 y.
d Decay product contribution is significant to determination of this value.

For the medium- and long-lived indicator radionuclides (C-14, Tc-99, U-238, Pu-239, and Am-241 [as Np-237]), the permissible waste concentrations acceptable for disposal in the generic tumulus are approximately twice those for the generic trench (within round off error). The performance for these radionuclides is based on site conditions after all engineered barriers within the disposal facilities have failed. At that time, the only assumed difference in the disposal facilities is their unit capacity; the generic trench is assumed to contain twice as much waste as the generic tumulus per unit volume.

For the short-lived radionuclides (H-3, Sr-90, and Cs-137), the permissible waste concentrations, where limited, are orders of magnitude higher for the generic tumulus than for the generic trench. These higher concentrations are due to the fact that the performance of the engineered barriers within the generic tumulus is assumed to be better than in the generic trench. As a result, the radionuclides are detained for a much longer period of time in the tumulus. The primary difference in the estimates for the limiting concentrations is directly due to differences in the time available for radioactive decay.

In general, based on technical considerations, most sites categorized as arid have much higher permissible waste concentrations than the humid sites for short-lived radionuclides and slightly higher permissible waste concentrations for long-lived radionuclides. This is due primarily to the radioactive decay that occurs during the longer travel times to the performance boundary at the arid sites (see Table 7-3).

Some radionuclides listed in Table 7-4 have no limit (NL) on their permissible waste concentration. These unlimited concentrations are the result of the combined effects of their relatively short half-lives and the extremely long travel times to the performance boundary. "No limit" is defined as a permissible waste concentration that is greater than the specific activity of the pure elemental radionuclide.

Estimates are that many of the radionuclides will arrive at the performance boundary beyond 10,000 y. The estimated waste concentrations for these radionuclides are presented in Table 7-4 for information purposes only. Consistent with the current approach used in performance assessments for LLW disposal (Wood et al., 1994), estimates of the peak radionuclide concentrations for arrivals at the performance boundary beyond 10,000 y were not considered in determining the most restrictive disposal limit from among the several pathways evaluated.

Values for the various parameters needed for calculating the key factors in the water pathway for the eight indicator radionuclides in the generic tumulus for the 15 sites are presented in Table 7-5. These include the CRF_{Source}, which is radionuclide-specific; the CRF_{Water}, which is site-specific; the soil distribution coefficients, K_d; the radionuclide travel times; the radioactive decay term, r_{Decay}; and the permissible waste concentrations for the water pathway, $C_{W\text{-}Water}$. In the tables, t_{cv} includes the time from facility closure to arrival at the water table and consequently, includes the detention time of radionuclides in the disposal facility plus the travel time in the unsaturated zone. The site-estimated water travel time in the vadose zone at NTS (Frenchman Flats) far exceeds the performance period of 10,000 y; as a result, PE calculations for the water pathway were not performed for NTS (see Chapter 7 in Volume 3). The K_d values do not include any effects of chelating agents; these chelating agents are assumed not to be in the waste.

Table 7-5. Input Parameters and Estimated Waste Concentrations for the Water Pathway at the 15 Sites (tumulus design) (Part 1 of 4)

(a) H-3 ($CRF_{Source} = 0.9$)

	Arid Sites								Humid Sites						
Parameter	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	5800 [c]	5.1	4.7
K_d (mL/g)[b]	0.1	0	-	0	0.1	0.1	0/0.1	0.1	0.1	0.1	0.1	0.1	0	0	0
t_{cv} (y)	2.7E+03	4.3E+02	-	4.2E+02	3.6E+02	2.1E+03	2.8E+03	6.3E+03	4.3E+02	3.5E+02	3.2E+02	3.2E+02	3.0E+01	3.1E+02	3.0E+01
t_{cs} (y)	4.6	1.4	-	1.8E-01	4.0E+01	9.6E+01	2.1	7.4	5	1.8	1.6	6.5	1.2E+00	3.7E+00	9.4E+02
t_{total} (y)[d]	2.7E+03	4.3E+02	-	4.2E+02	4.0E+02	2.3E+03	2.8E+03	6.3E+03	4.3E+02	3.5E+02	3.2E+02	3.2E+02	3.1E+01	3.1E+02	9.7E+02
r_{decay}	>1.0E+50	3.3E+10	-	1.8E+10	7.6E+09	>1.0E+50	>1.0E+50	>1.0E+50	3.4E+10	3.4E+08	6.9E+07	8.2E+07	5.8E+00	4.2E+07	4.4E+23
$C_{W\text{-}Water}$ (μCi/m³)	NL [e]	1E+14	-	3E+14	2E+12	NL	NL	NL	1E+13	1E+12	7E+10	3E+10	3E+06	2E+10	NL

(b) C-14 ($CRF_{Source} = 54$)

	Arid Sites								Humid Sites						
Parameter	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	2.0	5.1	4.7
K_d (mL/g)[b]	5	0	-	0	5	5	0 / 5	5	5	5	5	5	0	2	1
t_{cv} (y)	6.6E+04	4.3E+02	-	4.2E+02	2.0E+03	6.8E+04	8.4E+04	2.0E+05	3.7E+03	1.5E+03	7.2E+02	7.0E+02	3.0E+02	5.3E+02	3.0E+01
t_{cs} (y)	1.0E+02	1.4	-	1.8E-01	1.2E+03	1.9E+03	4.2E+01	1.7E+02	2.3E+02	4.1E+01	3.0E+01	1.0E+02	1.2E+00	6.9E+01	4.1E+03
t_{total} (y)[d]	6.6E+04	4.3E+02	-	4.2E+02	3.2E+03	7.0E+04	8.4E+04	2.0E+05	3.9E+03	1.5E+03	7.5E+02	8.0E+02	3.0E+02	6.0E+02	4.1E+03
r_{decay}	3.0E+03	1.1E+00	-	1.1E+00	1.5E+00	4.9E+03	2.6E+04	3.5E+10	1.6E+00	1.2E+00	1.1E+00	1.1E+00	1.0E+00	1.1E+00	1.6E+00
$C_{W\text{-}Water}$ (μCi/m³)	4E+07 [f]	7E+03	-	3E+04	8E+02	6E+06 [f]	8E+08 [f]	NL [f]	1E+03	7E+03	2E+03	7E+02	3E+02	8E+02	1E+03 [f]

a Water pathway was not evaluated for this site.

b Where only a single value is presented, it is for both the unsaturated and saturated zones. Where two numbers are given, the first is for the unsaturated zone and the second is for the saturated zone.

c Based on the infiltration rate through the facility derived from estimates of the performance of concrete vault boxes; see Chapter 5 in this volume for details.

d Sum of t_{vadose} and t_{cs}

e No Limit - the estimated concentration is higher than the specific activity of the pure elemental radionuclide.

f Radionuclide concentration is based on an arrival time beyond 10,000 y.

Table 7-5. Input Parameters and Estimated Waste Concentrations for the Water Pathway at the 15 Sites (tumulus design) (Part 2 of 4)

(c) Sr-90 ($CRF_{Source} = 540$)

	Arid Sites								Humid Sites						
Parameter	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	2.0	5.1	4.7
K_d (mL/g)[b]	15	1	-	2	15	15	116 / 15	15	15	15	20 / 15	15	30	10	5
t_{cv} (y)	2.0E+05	2.6E+03	-	2.6E+03	5.5E+03	2.0E+05	1.3E+06	6.0E+05	1.0E+04	3.7E+03	2.0E+03	1.4E+03	3.0E+01	1.5E+03	5.0E+01
t_{cs} (y)	3.1E+02	8.7	-	7.0E+00	3.4E+03	5.5E+03	1.2E+02	4.9E+02	6.8E+02	1.2E+02	8.7E+01	4.0E+02	1.7E+03	3.3E+02	1.7E+04
t_{total} (y)[d]	2.0E+05	2.6E+03	-	2.6E+03	8.9E+03	2.1E+05	1.3E+06	6.0E+05	1.1E+04	3.7E+03	2.1E+03	1.8E+03	1.7E+03	1.8E+03	1.7E+04
r_{decay}	>1.0E+50	7.5E+26	-	3.8E+26	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	4.9E+38	1.6E+21	1.8E+18	2.9E+17	1.8E+18	>1.0E+50
$C_{W\text{-}Water}$ (μCi/m³)	NL [ef]	NL	-	NL	NL	NL [f]	NL [f]	NL [f]	NL [f]	NL	NL	NL	NL	NL	NL [f]

(d) Tc-99 ($CRF_{Source} = 6.2$)

	Arid Sites								Humid Sites						
Parameter	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	2.0	5.1	4.7
K_d (mL/g)[b]	0.1	0	-	0.15	0.1	0.1	0.3 / 0.1	0.1	0.1	0.1	0.02	0.1	0	0.4	0 / 4
t_{cv} (y)	2.7E+03	4.3E+02	-	5.8E+02	3.6E+02	2.2E+03	5.6E+03	6.3E+03	4.3E+02	3.5E+02	3.1E+02	3.1E+02	3.0E+02	3.4E+02	3.0E+01
t_{cs} (y)	4.6	1.4	-	6.9E-01	4.0E+01	9.6E+01	2.1	7.4E+00	5	1.8	1.2	6.5	1.2	1.5E+01	9.1E+03
t_{total} (y)[d]	2.7E+03	4.3E+02	-	5.8E+02	4.0E+02	2.3E+03	5.6E+03	6.3E+03	4.3E+02	3.5E+02	3.1E+02	3.2E+02	3.0E+02	3.6E+02	9.1E+03
r_{decay}	1.0	1.0	-	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0	1.0
$C_{W\text{-}Water}$ (μCi/m³)	2E+03	1E+03	-	5E+03	9E+01	2E+02	5E+03	5E+03	1E+02	9E+02	3E+02	1E+02	5E+01	1E+02	1E+02

a Water pathway was not evaluated for this site.

b Where only a single value is presented, it is for both the unsaturated and saturated zones. Where two numbers are given, the first is for the unsaturated zone and the second is for the saturated zone.

c Based on the infiltration rate through the facility derived from estimates of the performance of concrete vault boxes; see Chapter 5 in this volume for details.

d Sum of t_{vadose} and t_{cs}

e No Limit - the estimated concentration is higher than the specific activity of the pure elemental radionuclide.

f Radionuclide concentration is based on an arrival time beyond 10,000 y.

Table 7-5. Input Parameters and Estimated Waste Concentrations for the Water Pathway at the 15 Sites (tumulus design) (Part 3 of 4)

(e) Cs-137 (CRF_{Source} = 54)

Parameter	Arid Sites								Humid Sites						
	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	2.0	5.1	4.7
K_d (mL/g)[b]	280	1	-	20	280	280	428 / 280	280	280	280	280	280	3000	100	40
t_{cv} (y)	3.6E+06	2.6E+03	-	2.2E+04	9.6E+04	3.8E+06	8.7E+06	1.1E+07	1.9E+05	4.2E+03	2.3E+04	2.1E+04	3.0E+01	1.2E+04	3.0E+01
t_{cs} (y)	5.7E+03	8.7	-	6.8E+01	6.4E+04	1.0E+05	2.3E+03	9.1E+03	1.3E+04	2.2E+03	1.6E+03	5.5E+03	1.4E+05	3.3E+03	1.3E+05
t_{total} (y)[d]	3.6E+06	2.6E+03	-	2.2E+04	1.6E+05	3.9E+06	8.7E+06	1.1E+07	2.0E+05	6.4E+03	2.5E+04	2.7E+04	1.4E+05	1.5E+04	1.3E+05
r_{decay}	>1.0E+50	7.9E+25	-	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50	>1.0E+50
$C_{W\text{-}Water}$ (µCi/m³)	NL [ef]	NL	-	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]	NL [f]

(f) U-238 (CRF_{Source} = 540)

Parameter	Arid Sites								Humid Sites						
	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	2.0	5.1	4.7
K_d (mL/g)[b]	35	0	-	1000 / 100	35	35	1.8 / 35	35	35 / 7	35	3.1 / 1.68	35	40	50	35 / 9
t_{cv} (y)	4.6E+05	4.3E+02	-	1.1E+06	1.2E+04	4.8E+05	6.0E+05	1.4E+06	2.4E+04	8.0E+03	5.7E+02	3.0E+03	3.0E+02	5.9E+03	3.0E+01
t_{cs} (y)	7.1E+02	1.4	-	3.4E+02	8.0E+03	1.3E+04	2.8E+02	1.1E+03	3.2E+02	2.8E+02	1.1E+01	6.9E+02	2.2E+03	1.7E+03	5.9E+04
t_{total} (y)[d]	4.6E+05	4.3E+02	-	1.1E+06	2.0E+04	4.9E+05	6.0E+05	1.4E+06	2.4E+04	8.3E+03	5.8E+02	3.7E+03	2.2E+03	7.6E+03	5.9E+04
r_{decay}	NA [g]	NA	-	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
$C_{W\text{-}Water}$ (µCi/m³)	9E+02 [f]	5E+02	-	2E+03 [f]	4E+01 [f]	9E+01 [f]	2E+03 [f]	2E+03 [f]	5E+01 [f]	4E+02	1E+02	5E+01	2E+01	5E+01	5E+01 [f]

a Water pathway was not evaluated for this site.

b Where only a single value is presented, it is for both the unsaturated and saturated zones. Where two numbers are given, the first is for the unsaturated zone and the second is for the saturated zone.

c Based on the infiltration rate through the facility derived from estimates of the performance of concrete vault boxes; see Chapter 5 in this volume for details.

d Sum of t_{vadose} and t_{cs}

e No Limit - the estimated concentration is higher than the specific activity of the pure elemental radionuclide.

f Radionuclide concentration is based on an arrival time beyond 10,000 y.

g Not Applicable - radioactive decay is implicit in pathway dose conversion factor.

Table 7-5. Input Parameters and Estimated Waste Concentrations for the Water Pathway at the 15 Sites (tumulus design) (Part 4 of 4)

(g) Pu-239 ($CRF_{Source} = 540$)

	Arid Sites								Humid Sites						
Parameter	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	2.0	5.1	4.7
K_d (mL/g)[b]	550	100	-	2000 / 200	550	550	110 / 550	550	550 / 500	550	550	550	40	100	550 / 1760
t_{cv} (y)	7.1E+06	2.2E+05	-	2.1E+06	1.8E+05	7.4E+06	1.0E+07	2.2E+07	3.7E+05	1.0E+05	4.6E+04	4.3E+04	3.0E+02	1.2E+04	3.0E+01
t_{cs} (y)	1.1E+04	7.3E+02	-	6.8E+02	1.3E+05	2.0E+05	4.5E+03	1.8E+04	2.3E+04	3.4E+04	3.2E+03	1.1E+04	1.9E+03	3.3E+03	4.2E+06
t_{total} (y)[d]	7.1E+06	2.2E+05	-	2.1E+06	3.1E+05	7.6E+06	1.0E+07	2.2E+07	3.9E+05	1.3E+05	4.9E+04	5.4E+04	2.2E+03	1.5E+04	4.2E+06
r_{decay}	>1.0E+50	5.2E+02	-	6.0E+26	8.5E+03	>1.0E+50	>1.0E+50	>1.0E+50	7.8E+04	3.7E+01	4.1E+00	4.7E+00	1.1E+00	1.5E+00	>1.0E+50
$C_{W\text{-}Water}$ ($\mu Ci/m^3$)	NL [e,f]	2E+04 [f]	-	NL [f]	3E+04 [f]	NL [f]	NL [f]	NL [f]	3E+05 [f]	1E+03 [f]	4E+01 [f]	2E+01 [f]	2E+00	6E+00 [f]	NL [f]

(h) Am-241 ($CRF_{Source} = 540$)

	Arid Sites								Humid Sites						
Parameter[g]	LLNL	Hanford	NTS[a]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Water}	88	45	-	190	4	8.5	210	220	4.5	39	12	4.5	2.0	5.1	4.7
K_d (mL/g)[b]	5	10	-	50 / 5	5	5	0.007 / 5	5	5 / 20	5	2.4	5	40	10	5
t_{cv} (y)	6.6E+04	2.2E+04	-	5.4E+04	2.2E+03	7.0E+04	8.4E+04	2.0E+05	4.4E+03	1.4E+03	5.2E+02	7.9E+02	3.0E+02	1.6E+03	3.0E+01
t_{cs} (y)	3.8E+01	7.3E+01	-	6.4E+00	1.0E+03	6.8E+02	1.5E+01	6.2E+02	2.3E+02	1.5E+03	1.3E+00	1.0E+01	1.9E+03	2.0E+02	6.0E+06
t_{total} (y)[d]	6.6E+04	2.2E+04	-	5.4E+04	3.2E+03	7.0E+04	8.4E+04	2.0E+05	4.6E+03	1.5E+03	5.2E+02	8.0E+02	2.2E+03	1.8E+03	6.0E+06
r_{decay}	1.0E+00	1.0E+00	-	1.0E+00	1.0E+00	1.0E+00	1.0E+00	1.0E+00	1.0E+00	1.0E+00	1.0E+00	1.0E+00	3.1E+01	1.0E+00	1.0E+00
$C_{W\text{-}Water}$ ($\mu Ci/m^3$)	3E+05 [f]	1E+05 [f]	-	6E+05 [f]	1E+04 [f]	3E+04 [f]	7E+05 [f]	8E+05 [f]	1E+04 [f]	1E+05	4E+04	1E+04	5E+01	2E+04	2E+04 [f]

a Water pathway was not evaluated for this site.

b Where only a single value is presented, it is for both the unsaturated and saturated zones. Where two numbers are given, the first is for the unsaturated zone and the second is for the saturated zone.

c Based on the infiltration rate through the facility derived from estimates of the performance of concrete vault boxes; see Chapter 5 in this volume for details.

d Sum of t_{vadose} and t_{cs}

e No Limit - the estimated concentration is higher than the specific activity of the pure elemental radionuclide.

f Radionuclide concentration is based on an arrival time beyond 10,000 y.

g Except for ORR where the dose is controlled by Am-241, K_d, travel times, and r_{decay} are those of Np-237, the dominant decay product of Am-241.

To simplify the discussion of the indicator radionuclides, only the results for the generic tumulus are presented. As discussed previously, the permissible waste concentrations for the generic trench and tumulus disposal technologies are similar for the long-lived radionuclides. Further, the short-lived radionuclides generally have more restrictive water pathway concentrations at humid sites, which are more likely to use tumulus technology. For example, ORR currently uses tumulus technology for LLW disposal and SRS uses a below-ground vault facility which has engineered barriers that are more robust than those for a tumulus facility. The performance of each indicator radionuclide at the 15 sites is discussed below.

H-3

Values for the key input parameters and the estimated permissible concentrations for H-3 are shown in Table 7-5(a). Tritium is highly mobile with a short (12.3 y) half-life. As a result, sites with thick vadose zones and/or low recharge will provide travel times long enough for this radionuclide to decay to an insignificant level. These sites include LLNL, NTS, SNL, LANL, Pantex, and WVDP. As a result, there is no limit on the permissible concentrations of H-3 to be placed in disposal facilities at these four sites. In contrast, conditions at a humid site can be very different. Because of its high mobility, the amount of H-3 that can be leached from grouted waste in the generic tumulus at sites located in regions that are humid and have a relatively thin vadose zone can be significant even prior to failure of the engineered barriers (see Figure 5-3 of Volume 2). This early release is important in estimating permissible waste concentrations for short-lived, mobile radionuclides which would otherwise decay to insignificant levels in the tumulus prior to failure (at 300 y) of the engineered barriers. This early release is even more significant for the trench design because a higher release rate is assumed between 30 and 100 y after facility closure (see Figure 5-3 of volume 2). Consequently, as shown in Table 7-3, the permissible waste concentrations for H-3 are much lower for a trench than for a tumulus design.

In general, the arid sites have high (often unlimited) permissible waste concentrations for H-3 for the tumulus design. The humid sites have relatively high permissible waste concentrations for H-3 for the tumulus design.

C-14

Values for the key input parameters and the estimated permissible concentrations for C-14 are shown in Table 7-5(b). Carbon-14 is highly mobile with a half-life of 5700 y. The permissible concentrations for this radionuclide at LLNL, SNL, LANL, and Pantex are relatively high, and the arrival time at the performance boundary for these sites is beyond 10,000 y. Because of the general absence of water at NTS, it can also be assumed to have essentially an unlimited permissible waste concentration acceptable for disposal at that site. The travel times for C-14 at the remaining sites are relatively short. Therefore, significant radioactive decay of C-14 does not occur before it reaches the performance boundary.

Sr-90

Values for the key input parameters and the estimated permissible concentrations for Sr-90 are shown in Table 7-5(c). Because of its short half-life (29.1 y) and moderate sorption capacity, this radionuclide decays to an insignificant level (large r_{Decay}) at all evaluated sites prior

to reaching the performance boundary. Therefore, based on the results of the PE, the disposal of Sr-90 is not limited by the water pathway at any of the 15 sites.

Tc-99

Values for the key input parameters and the estimated permissible concentrations for Tc-99 are shown in Table 7-5(d). Because of its long half-life (2.1E5 y) and high mobility, this radionuclide does not undergo significant decay prior to reaching the performance boundary at any of the 15 sites. As a result, once Tc-99 migrates from the facility, its concentration is attenuated only by dilution of the leachate by groundwater. At ORR, in particular, the large recharge relative to groundwater flow results in a low permissible concentration.

Only NTS has a Tc-99 arrival time at the performance boundary greater than 10,000 y, due mainly to low or non-existent recharge and thick vadose zones. While permissible waste concentrations are uniformly low at all sites, the arid site concentrations are generally higher than those of the humid sites.

Cs-137

Values for the key input parameters and the estimated permissible concentrations for Cs-137 are shown in Table 7-5(e). Because of its short half-life (30.2 y) and high sorption capacity, Cs-137 decays to an insignificant level at all sites prior to reaching the performance boundary. Based on the results of the PE, Cs-137 disposal is not limited by the water pathway at any of the sites.

U-238

Values for the key input parameters and the estimated permissible concentrations for U-238 are shown in Table 7-5(f). This radionuclide has an extremely long half-life (4.5E9 y) and moderately low mobility. Therefore, the estimated permissible concentrations for U-238 are relatively low at all sites, even though eight sites have arrival times at the performance boundary in excess of 10,000 y. At Hanford, the mobility of U-238 through the soil is assumed to be high (i.e., low K_d values) so that even with the low infiltration rate and thick vadose zone, the radionuclide arrives at the performance boundary in less than 10,000 y. The soil at FEMP exhibits similar low K_d values with the result being that U-238 would also be anticipated to reach the performance boundary at this site in less than 10,000 y. The combination of moderate mobility, higher infiltration rates, and thinner vadose zones cause U-238 to arrive at the performance boundary before 10,000 y at Hanford, PGDP, FEMP, PORTS, ORR, and SRS.

Estimates of the transport of U-238 in the groundwater show that, for the arid sites, its movement will be sufficiently delayed that, in general, its arrival at the performance boundary will be in excess of the 10,000-y performance period. However, because of the extremely long half-life of the radionuclide (4.5 billion y), there could be significant concentrations in the groundwater regardless of when it arrives (i.e., even if the time is well beyond 10,000 y).

Pu-239

Values for the key input parameters and the estimated permissible concentrations for Pu-239 are shown in Table 7-5(g). This radionuclide has a long half-life (2.41E4 y) and generally

has low mobility. At most sites, the Pu-239 arrival times at the performance boundary are well beyond 10,000 y. At LLNL, NTS, INEL, SNL, LANL, Pantex, and WVDP, the combination of long groundwater travel times and low mobility causes estimated Pu-239 arrival times at the performance boundary from 2 million to 20 million years. These arrival times result in an unlimited permissible concentration for Pu-239 in wastes acceptable for disposal at these sites. At ORR and SRS where it is assumed to have medium mobility and a short water travel time, Pu-239 has a very low permissible waste concentration; however, at SRS this radionuclide arrives at the performance boundary beyond 10,000 y.

Am-241

Values for the key input parameters and the estimated permissible concentrations for Am-241 are shown in Table 7-5(h). The half-life of this radionuclide is 430 y but it decays to Np-237, which has a half-life of over two million years. Americium-241 generally has very low environmental mobility and therefore decays to insignificant levels prior to reaching the performance boundary. The exception is at ORR, where the assumption of much greater mobility, coupled with short water travel times, causes it to arrive at the performance boundary in an estimated time of 2200 y. As a result, the permissible concentration for Am-241 in wastes acceptable for disposal at that site is low.

Upon decay, Am-241 forms Np-237 which can be a significant source of exposure. In fact, for times beyond about 5,000 y, the estimated dose will be due primarily to Np-237. Even so, this radionuclide, which is mobile in the environment, will arrive at the performance boundary at times beyond 10,000 y at all sites except PGDP, FEMP, PORTS, ORR, and SRS.

7.3 RESULTS OF THE ATMOSPHERIC PATHWAY ANALYSIS

The atmospheric pathway analysis used in the PE is based on the use of a generic conceptual model that incorporates generic and site-specific data to estimate the transport of volatile radionuclides. Only two volatile radionuclides, H-3 and C-14, were analyzed for the atmospheric pathway. Other radionuclides, including I-129 and Cs-137, could become volatile under high temperature conditions but these conditions are not expected to be present at any of the disposal facilities. Because the analysis is quite generic, a common set of assumptions was used for the generic trench and tumulus design. For this reason, on the basis of the atmospheric pathway analysis there is no difference in the estimates of permissible waste concentrations for these two types of disposal facilities.

The attenuation in the radionuclide concentrations between the disposal facility and the performance boundary is represented by the soil diffusion concentration reduction factor, CRF_{Diff}, the atmospheric dispersion concentration reduction factor, CRF_{Disp}, and the radioactive decay term, r_{Decay}. The CRF_{Diff} represents the attenuation in the radionuclide concentrations between the disposed waste and the top of the soil surface. The CRF_{Disp} represents the attenuation between the top of the soil surface and the 100-m performance boundary. The r_{Decay} represents the radioactive decay that occurs prior to the arrival of the radionuclide at the performance boundary and includes 100 y of assumed detention in the disposal facility. The 100-y detention time is based on the following generic assumptions of the PE:

- The waste form was grouted MLLW treatment residuals. Based on this assumption, tritium as vapor was bound in the pore water of the hydrophilic grout, and carbon dioxide as a gas carrying the C-14 isotope was limited by the high pH of the grout, so that the waste form provided retention of these volatile radionuclides in the disposal facility.
- The disposal facility was capped by a RCRA-compliant cover system. Based on this assumption, the cover system was maintained to provide low permeability for 100 y.

The *CRF*s for soil diffusion and atmospheric dispersion, the radionuclide decay term, r_{Decay}, and the permissible concentrations for the atmospheric pathway for H-3 and C-14 are shown in Table 7-6. Comparison of the data for the 15 sites shows that the concentration reduction factors for H-3 and C-14 are similar, with CRF_{Diff} providing the largest concentration reduction for both radionuclides. Because the waste is assumed to begin migrating at the end of the 100-y institutional control period, the main difference between H-3 and C-14 comes from the r_{Decay} term; at 100 y H-3 has decayed significantly because of its short half-life while C-14 has decayed very little.

The important conclusions from the PE model for the atmospheric pathway are that (1) there is no significant difference in the permissible waste concentrations of volatile radionuclides acceptable for disposal at the 15 sites, even though site-specific soil and meteorological data were used in the calculations, and (2) the estimated permissible waste concentration limits for H-3 are much higher than C-14 because of the short half-life of H-3.

7.4 RESULTS OF THE ANALYSIS OF INADVERTENT INTRUSION SCENARIOS

The PE used two inadvertent intrusion (the "homesteader" and "post-drilling") scenarios based primarily on the assumptions used by the NRC staff in developing the waste classification system for near-surface disposal of radioactive waste in 10 CFR Part 61 (NRC, 1982). Because the behavior of future societies is difficult to predict, specific assumptions are made in this analysis to enable predictions of permissible radionuclides in waste.

The homesteader scenario is based on the assumption that, at some period after institutional control, an intruder establishes a permanent homestead above a closed disposal facility and develops on-site sources of water and agricultural products. The assumed time of intrusion was based on the disposal technology and site-specific factors. Intrusion into a trench facility was assumed to occur at 300 y, and intrusion into a tumulus facility was assumed to occur at 500 y. Waste in the disposal units is assumed to be accessed when the intruder constructs a home directly on top of the disposal facility and the foundation of the home extends into the waste. Some of the waste exhumed from the disposal facility then is assumed to be mixed with uncontaminated native soil in the intruder's vegetable garden. This mixing results in a chronic exposure through ingestion of vegetables and soil, external exposure, and inhalation. As discussed in Section 5.3.3, the homesteader scenario does not consider exposure from contaminated water. This simplification is based on results from LLW disposal performance assessments (MMES et al., 1994; ORNL, 1994). In these performance assessments, the times of maximum exposure via direct exposure and exposure from contaminated water are different such that the total dose is not the sum of the maximum doses for the two pathways.

Table 7-6. Concentration Reduction Factors, Radioactive Decay Terms, and Permissible Waste Concentrations for H-3 and C-14 for the Atmospheric Pathway at 15 Sites (tumulus and trench designs)

(a) H-3

	Arid Sites								Humid Sites						
Parameter	LLNL	Hanford	NTS	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Diff}	2E+06	8E+05	8E+05	4E+06	2E+06	8E+05	5E+05	2E+06	3E+06	2E+06	3E+06	2E+06	4E+06	4E+06	4E+06
CRF_{Disp}	1E+03	8E+02	1E+03	1E+03	1E+03	1E+03	1E+03	2E+03	3E+03	1E+03	8E+02	1E+03	5E+02	2E+03	6E+02
r_{Decay}	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02	3E+02
$C_{W\text{-}Atm}$ ($\mu Ci/m^3$)	**7E+09**	**2E+09**	**3E+09**	**1E+10**	**7E+09**	**3E+09**	**2E+09**	**1E+10**	**3E+10**	**7E+09**	**7E+09**	**7E+09**	**7E+09**	**3E+10**	**9E+09**

(b) C-14

	Arid Sites								Humid Sites						
Parameter	LLNL	Hanford	NTS	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
CRF_{Diff}	4E+04	2E+04	2E+04	8E+04	4E+04	2E+04	1E+04	4E+04	7E+04	5E+04	6E+04	4E+04	9E+04	1E+05	8E+04
CRF_{Disp}	2E+03	1E+03	2E+03	2E+03	2E+03	2E+03	2E+03	3E+03	4E+03	2E+03	1E+03	2E+03	1E+03	4E+03	1E+03
r_{Decay}	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00	1E+00
$C_{W\text{-}Atm}$ ($\mu Ci/m^3$)	**4E+03**	**1E+03**	**2E+03**	**7E+03**	**4E+03**	**2E+03**	**1E+03**	**5E+03**	**1E+04**	**5E+03**	**3E+03**	**4E+03**	**4E+03**	**2E+04**	**4E+03**

The post-drilling scenario is based on the assumption that an intruder who resides permanently near the disposal site drills through a disposal cell while constructing a well for a domestic water supply. The contaminated drill cuttings brought to the surface are assumed to be mixed with uncontaminated native soil in the intruder's vegetable garden. In this case, the relevant pathways for chronic exposure include ingestion of vegetables grown in the contaminated garden soil, ingestion of contaminated soil, and external and internal exposure, the latter occurring primarily through inhalation, while working in the garden. Again, the exposure to contaminated water is not considered for the same reasons given for the homesteader scenario.

The homesteader scenario is assumed to occur at 300 y after closure for the generic trench and at 500 y for the generic tumulus. The waste form is assumed to be indistinguishable from its surroundings at these times. Additional credit is given to the tumulus facility because of the concrete vault-boxes incorporated into this design. The post-drilling scenario is assumed to occur after the 100-y institutional control period for both the trench and tumulus designs in hard-rock drilling regions when the waste is assumed indistinguishable from natural formations. In regions of soft-formation drilling (e.g., Savannah River Site), post-drilling intrusion is assumed not to occur until after 500 y for the tumulus and 300 y for the trench. The assumed lifetime of engineered barriers for preventing intrusion is consistent with that used by the NRC in developing the waste classification system in 10 CFR Part 61 (NRC, 1982). Exceptions to these times of intrusion are made for radionuclides that have increasing doses throughout the compliance period due to in-growth of decay products (e.g., the indicator radionuclide U-238). For these radionuclides, the time of intrusion is assumed to be 10,000 y in order to obtain the maximum dose within the period of performance.

The permissible waste concentrations for the indicator radionuclides at the 15 sites for the trench and tumulus designs are shown in Table 7-7. The post-drilling scenario generally yields more restrictive concentration limits than the homesteader scenario for those radionuclides with short half-lives (i.e., H-3, Sr-90 and Cs-137) except for Cs-137 in the trench design. Cesium-137 has a half-life that is just large enough for that radionuclide to be controlled by the homesteader scenario for the trench design. In general, the more restrictive concentration limits for the post-drilling scenario are due to the assumption that this scenario takes place 200 and 400 y earlier than the homesteader intrusion scenario for the trench and tumulus designs, respectively. In general, the differences in permissible waste concentrations for the intruder pathway between the trench design and the tumulus design are not very significant for long-lived radionuclides.

Table 7-7. Permissible Waste Concentrations ($\mu Ci/m^3$) for the Standard Intrusion Scenarios

Nuclide	Trench		Tumulus	
	$C_{W\text{-}Intr}$ Homesteader ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Post-Drilling ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Homesteader ($\mu Ci/m^3$)	$C_{W\text{-}Intr}$ Post-Drilling ($\mu Ci/m^3$)
H-3	1E12	***7E7***[a]	NL[b]	***7E7***
C-14	***1E4***	7E4	***1E4***	7E4
Sr-90	1E6	***5E4***	1E8	***5E4***
Tc-99	***2E4***	8E4	***2E4***	8E4
Cs-137	***3E5***	8E5	3E7	***8E5***
U-238	***5E3***	1E5	***5E3***	1E5
Pu-239	***6E3***	5E4	***6E3***	5E4
Am-241	***7E3***	5E4	***9E3***	5E4

a The limiting radionuclide concentrations for each facility design are highlighted in bold italics.
b No Limit–the estimated concentration is higher than the specific activity of the pure elemental radionuclide.

7.5 COMPARISON OF WATER AND ATMOSPHERIC PATHWAYS AND INTRUSION SCENARIOS

The trends in performance of the radionuclides for the three exposure pathways have been summarized in the previous sections using eight indicator radionuclides. In Section 7.5.1, the indicator radionuclides are used to summarize the general trends in the exposure pathways limiting the permissible waste concentrations at the 15 sites. After this general discussion using indicator radionuclides, a summary of the exposure pathways limiting the permissible waste concentrations based on the entire 58 radionuclides is presented in Section 7.5.2.

7.5.1 Indicator Radionuclides

The permissible concentrations for the eight indicator radionuclides for the generic tumulus at the 15 sites for the water and atmospheric pathways and the intrusion scenarios are shown in Table 7-8. For intrusion, only the more restrictive value from the homesteader and post-drilling scenario is shown. It should also be noted that the atmospheric pathway is applicable only to H-3 and C-14; as previously explained, no other radionuclides were evaluated for this pathway.

For the arid sites, the intruder pathway provides the limiting permissible concentrations for most of the indicator radionuclides. The atmospheric pathway is limiting for C-14 at the arid sites due in part to the conservative assumptions used in this pathway analysis. For the humid sites, the number of indicator radionuclides limited by the intruder scenarios is more comparable to the number limited by the water pathway. Technetium-99 is limited by the water pathway at all sites that have an arrival time at the performance boundary prior to 10,000 y.

Table 7-8. Summary of the Permissible Waste Concentrations ($\mu Ci/m^3$) for the Indicator Radionuclides (tumulus design)

		Arid Sites								Humid Sites						
		LLNL	Hanford	NTS	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP
H-3	Water	NL[a]	1E14	-[b]	3E14	2E12	NL	NL	NL	1E13	1E12	7E10	3E10	***3E6*** [c]	***2E10***	NL
	Atmospheric	7E9	2E9	3E9	1E10	7E9	3E9	2E9	1E10	3E10	7E9	7E9	7E9	7E9	3E10	9E9
	Intrusion	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	***7E7***	7E7	NL	***7E7***
C-14	Water	-	7E3	-	3E4	***8E2***	-	-	-	***1E3***	7E3	***2E3***	***7E2***	***3E2***	***8E2***	***1E3***
	Atmospheric	***4E3***	***1E3***	***2E3***	***7E3***	4E3	***2E3***	***1E3***	***5E3***	1E4	***5E3***	3E3	4E3	4E3	2E4	4E3
	Intrusion	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4	1E4
Sr-90	Water	-	NL	-	NL	NL	-	-	-	-	NL	NL	NL	NL	NL	-
	Intrusion	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***5E4***	***1E8***	***5E4***
Tc-99	Water	***2E3***	***1E3***	-	***5E3***	***9E1***	***2E2***	***5E3***	***5E3***	***1E2***	***9E2***	***3E2***	***1E2***	***5E1***	***1E2***	***1E2***
	Intrusion	2E4	2E4	***2E4***	2E4	2E4	2E4	2E4	2E4	2E4	2E4	2E4	2E4	2E4	2E4	2E4
Cs-137	Water	-	NL	-	-	-	-	-	-	-	-	-	-	-	-	-
	Intrusion	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***8E5***	***3E7***	***8E5***
U-238	Water	-	***5E2***	-	-	-	-	-	-	-	***4E2***	***1E2***	***5E1***	***2E1***	***5E1***	-
	Intrusion	***5E3***	5E3	***5E3***	***5E3***	***5E3***	***5E3***	***5E3***	***5E3***	***5E3***	5E3	5E3	5E3	5E3	5E3	***5E3***
Pu-239	Water	-	-	-	-	-	-	-	-	-	-	-	-	***2E0***	-	-
	Intrusion	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	***6E3***	6E3	***6E3***	***6E3***
Am-241 [d]	Water	-	-	-	-	1E4	-	-	-	1E4	1E5	4E4	1E4	***5E1***	2E4	-
	Intrusion	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	***9E3***	9E3	***9E3***	***9E3***

a No Limit - the estimated permissible waste concentration is larger than the specific activity of the pure elemental radionuclide.
b Indicates that the radionuclide arrival time at the performance boundary is beyond 10,000 y (water pathway was not evaluated for NTS).
c The most restrictive pathway is shown in bold italics.
d The decay product Np-237 contributes significantly to the results for all sites but ORR, which has an earlier arrival time.

The concentration limits for the indicator radionuclides, as specified by the NRC in Title 10 CFR Part 61 for disposal of commercial Class A LLW, are shown in Table 7-9. Limits are given for all the indicator radionuclides except U-238; the NRC waste classification system does not include a concentration limit for this radionuclide.

Table 7-9. NRC Class A Limits for Radionuclide Concentrations for the Indicator Radionuclides

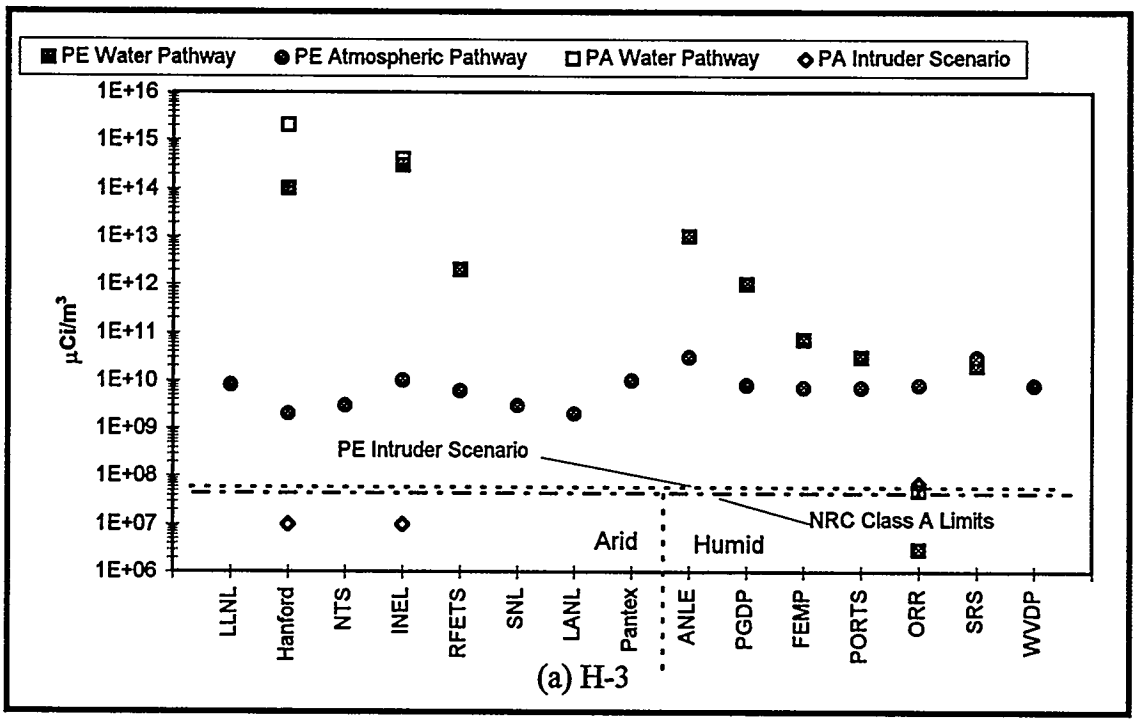
Radionuclide	Concentration Limits ($\mu Ci/m^3$)
H-3	4E+07
C-14	8E+05
Sr-90	4E+04
Tc-99	3E+05
Cs-137	1E+04
U-238	_ [b]
Pu-239[a]	2E+04
Am-241[a]	2E+04

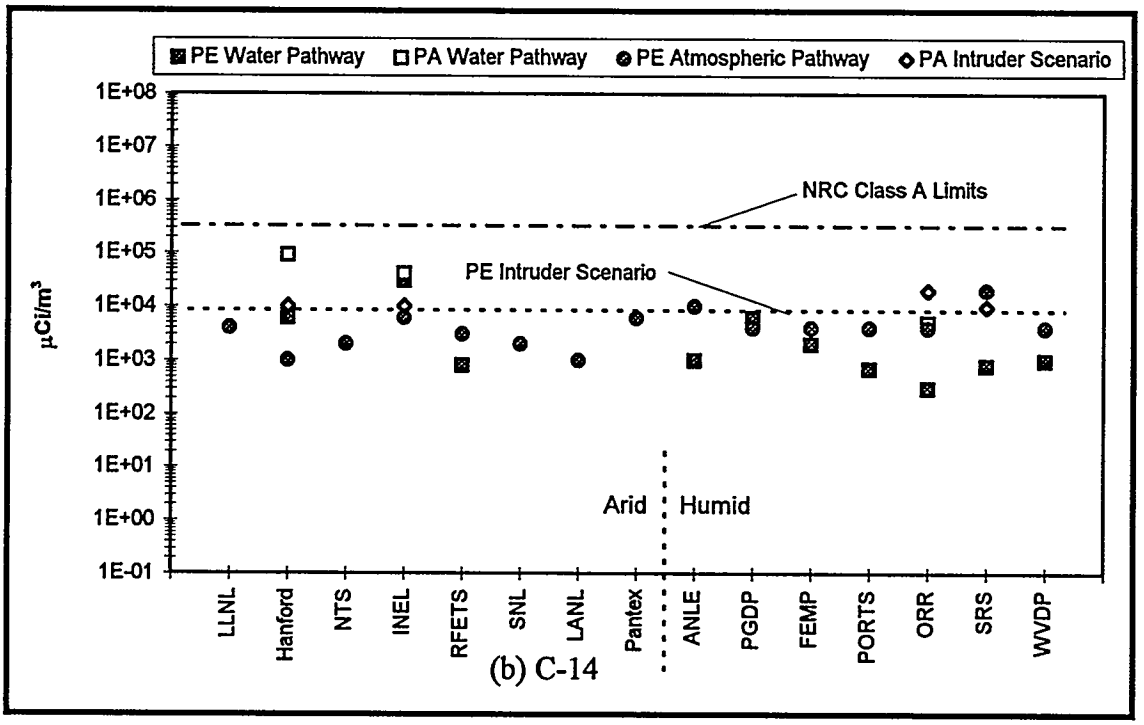
a Alpha emitting transuranic nuclide with half-life greater than 5 y based on 10 nCi/g limit assuming waste density of 2.4 g/cm^3

b The NRC waste classification system does not include a concentration limit for this radionuclide.

The permissible waste concentrations for the indicator radionuclides for the water pathway, the atmospheric pathway, and the intruder scenarios as analyzed in the PE; the water pathway and intrusion scenario results based on LLW performance assessments at four sites; and the NRC limits for the indicator radionuclides in Class A LLW are presented in Figure 7-3. Performance assessments for NTS and LANL are being drafted and are not available for detailed comparison. The values for the permissible waste concentrations from the performance assessments were developed by applying to the generic tumulus design the transport assumptions used in that performance assessment.

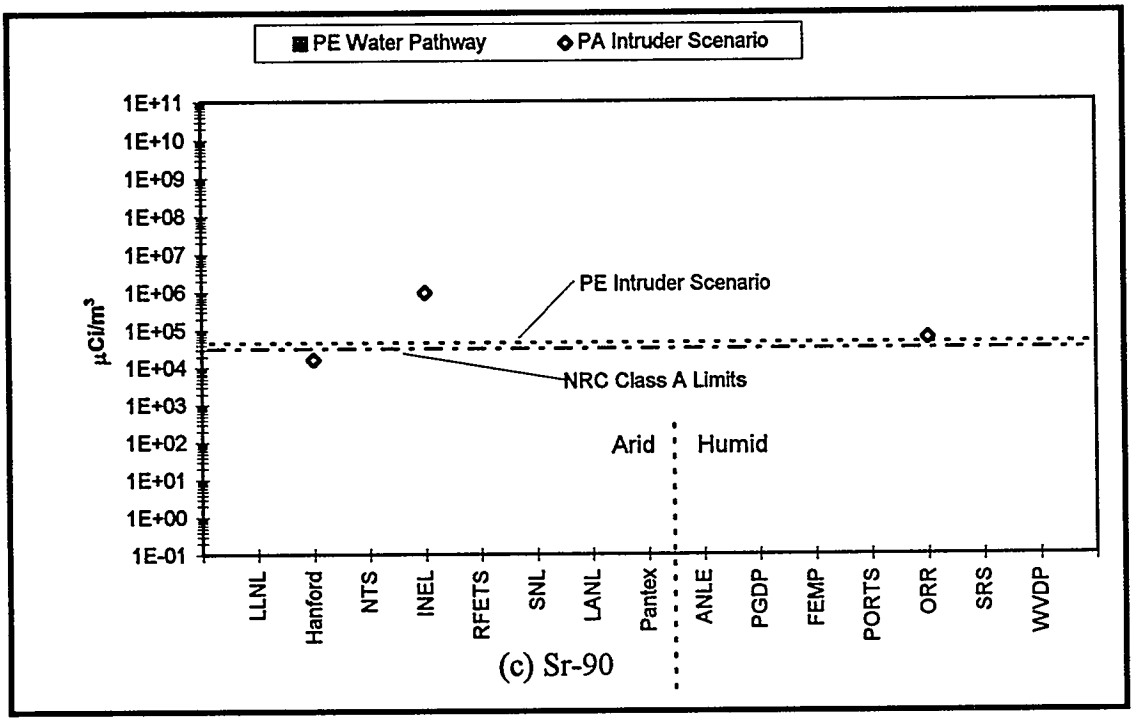
The permissible waste concentrations derived from the performance assessments and NRC waste classification system are presented to provide a comparison with the permissible waste concentrations based on the PE. Each indicator radionuclide is discussed below. The degree of comparison between the permissible waste concentrations as analyzed in the PE and the performance assessments is primarily due to differences in the assumptions used in the two analyses. These differences are discussed in Section 7.6.

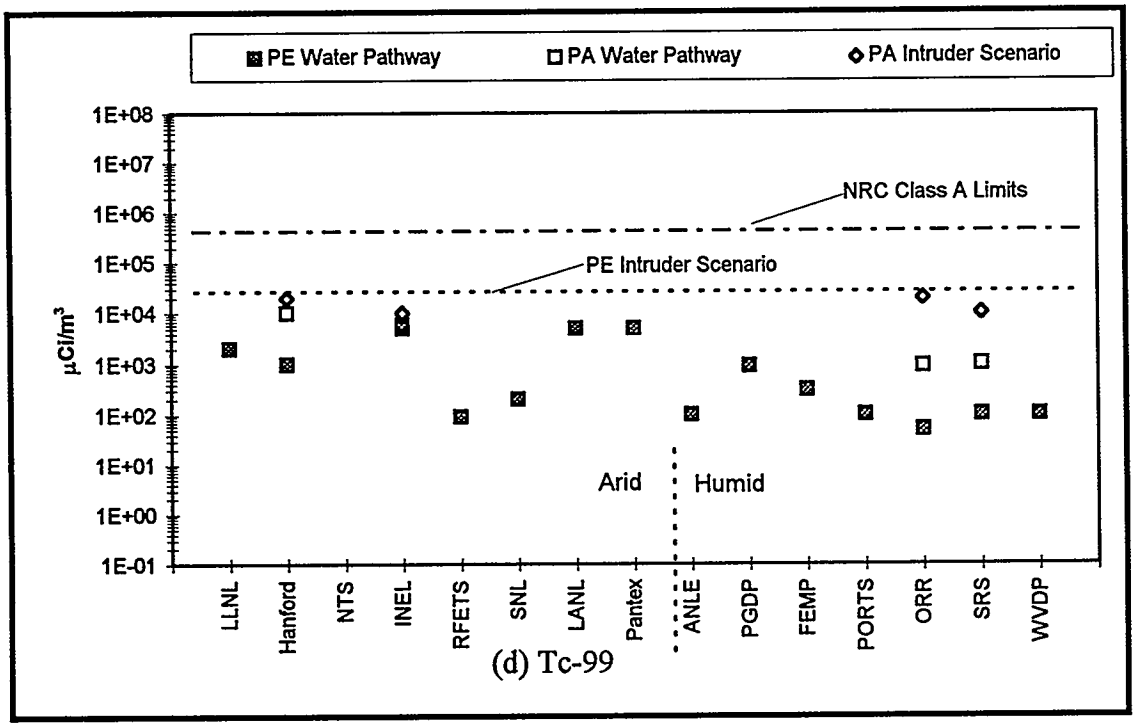




Note: Permissible radionuclide concentrations in the waste for the water pathway are presented for radionuclides that arrive at the performance boundary before 10,000 y.

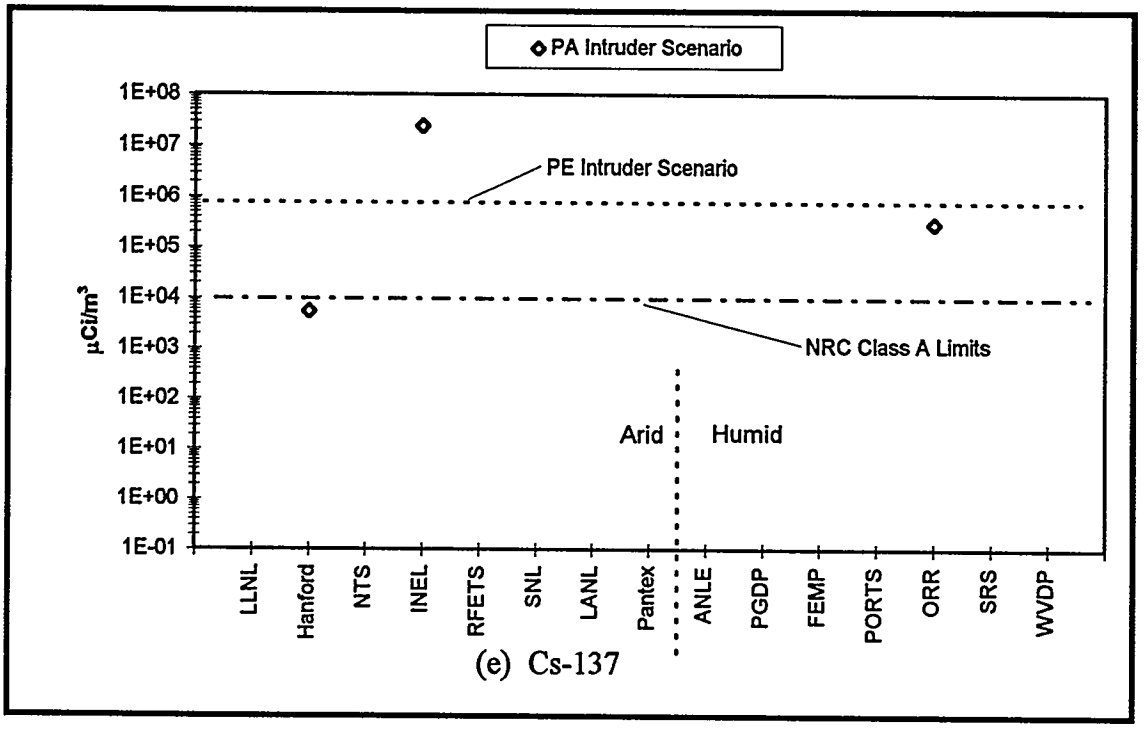
Figure 7-3. Generic tumulus permissible concentrations in waste for the indicator radionuclides compared with NRC limits for radionuclides in Class A wastes and the performance assessment results for four sites. (Part 1 of 4)

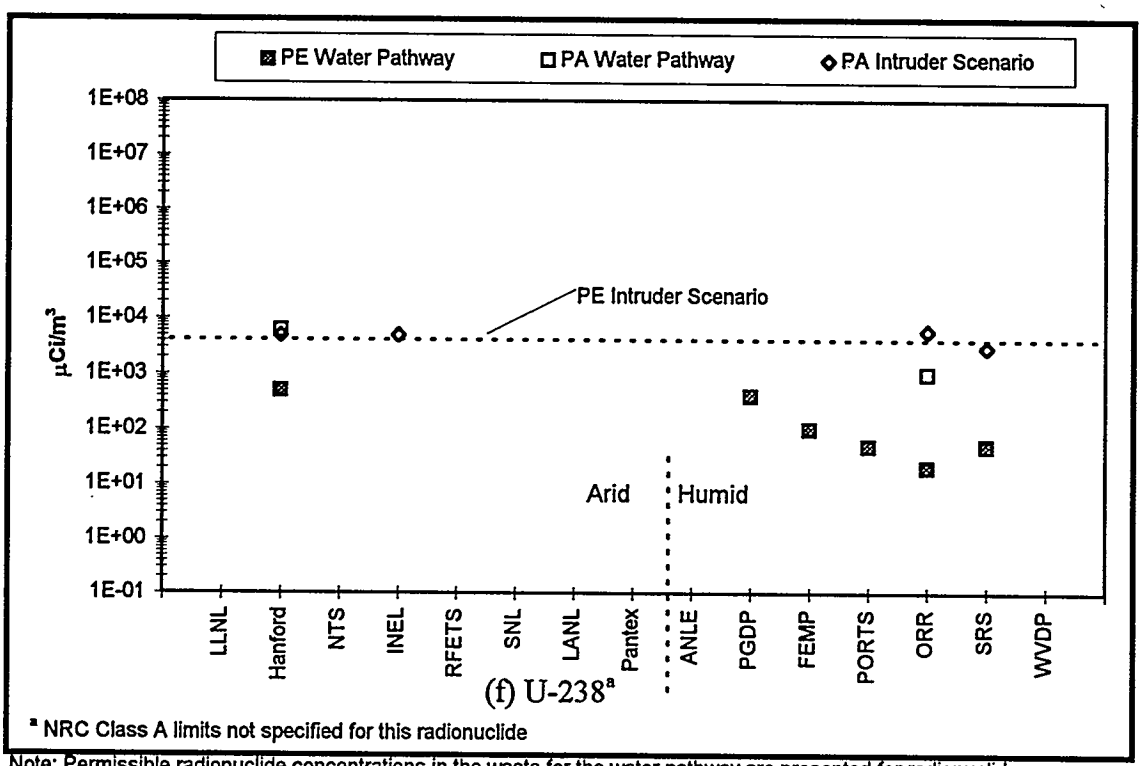




Note: Permissible radionuclide concentrations in the waste for the water pathway are presented for radionuclides that arrive at the performance boundary before 10,000 y.

Figure 7-3. Generic tumulus permissible concentrations in waste for the indicator radionuclides compared with NRC limits for radionuclides in Class A wastes and the performance assessment results for four sites. (Part 2 of 4)

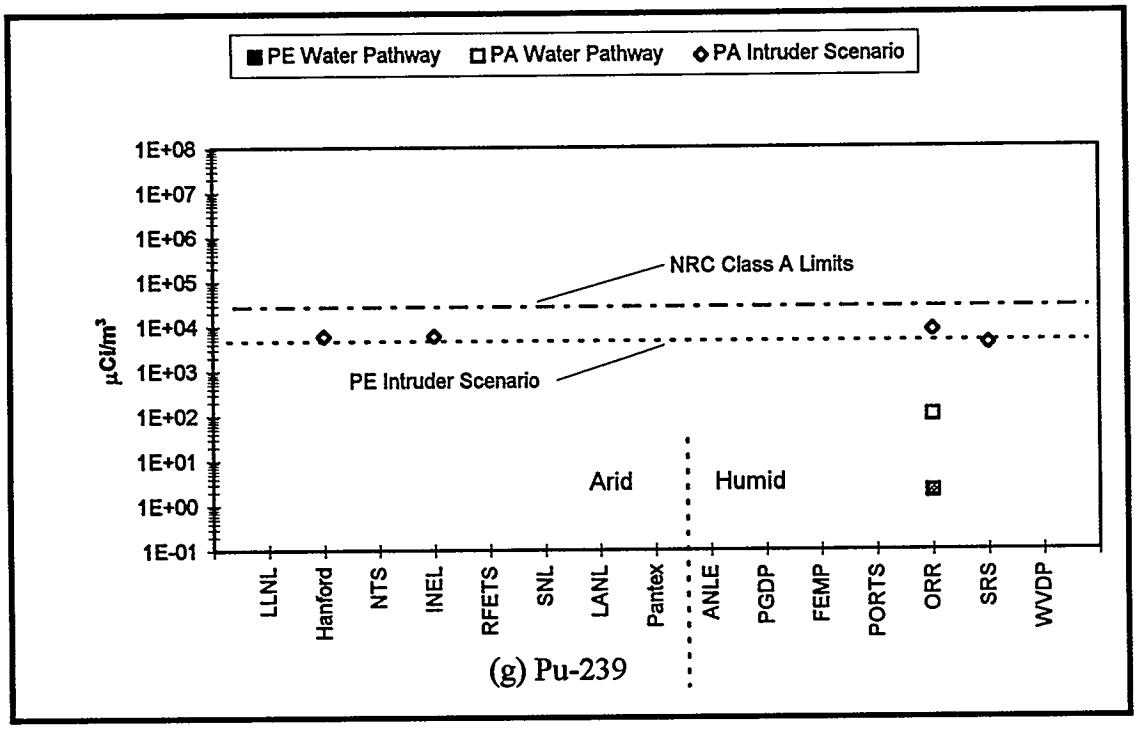


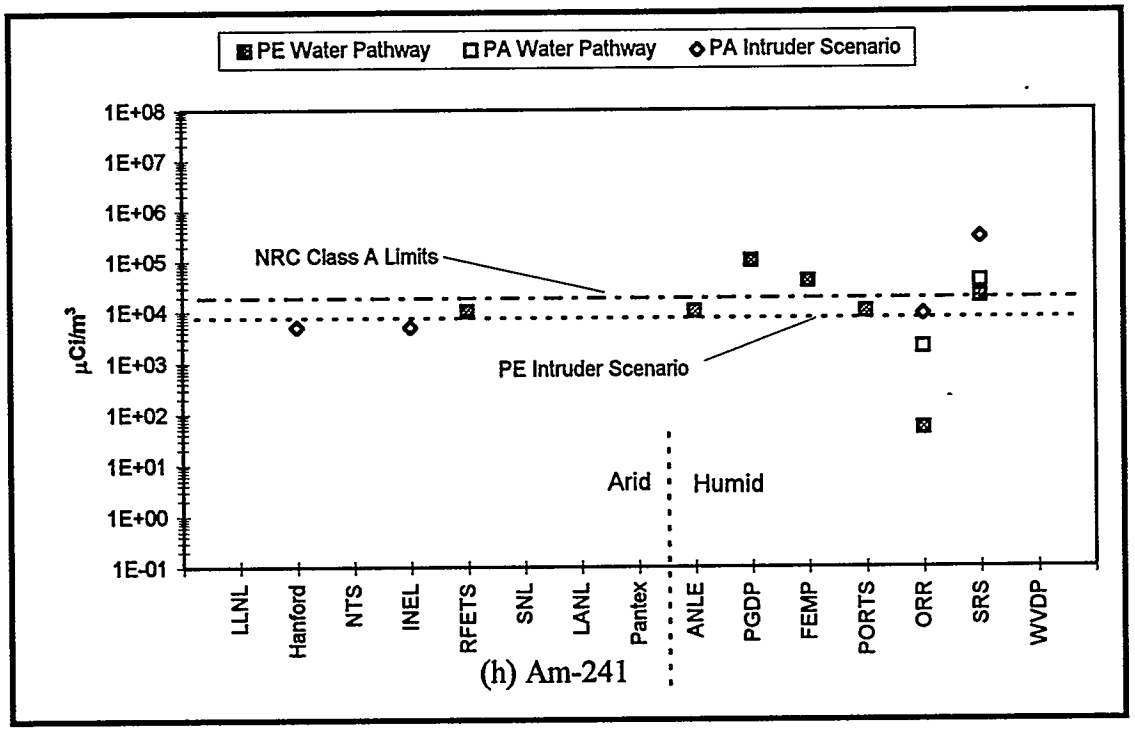


[a] NRC Class A limits not specified for this radionuclide

Note: Permissible radionuclide concentrations in the waste for the water pathway are presented for radionuclides that arrive at the performance boundary before 10,000 y.

Figure 7-3. Generic tumulus permissible concentrations in waste for the indicator radionuclides compared with NRC limits for radionuclides in Class A wastes and the performance assessment results for four sites. (Part 3 of 4)





Note: Permissible radionuclide concentrations in the waste for the water pathway are presented for radionuclides that arrive at the performance boundary before 10,000 y.

Figure 7-3. Generic tumulus permissible concentrations in waste for the indicator radionuclides compared with NRC limits for radionuclides in Class A wastes and the performance assessment results for four sites. (Part 4 of 4)

H-3

Comparisons of permissible waste concentrations estimated in the PEs and LLW performance assessments and the NRC limits in Class A LLW for H-3 are shown in Figure 7-3(a). The permissible concentrations for tritium, based on the PE, are limited by the intrusion pathway at all sites except ORR, where the water pathway is limiting. The atmospheric pathway concentrations are one to two orders of magnitude larger (less restrictive) than the intrusion concentrations. The water pathway concentrations are even higher, with several sites (LLNL, NTS, SNL, LANL, Pantex, and WVDP) having no limit for this radionuclide for this pathway due to its relatively rapid decay (Table 7-8). As may be noted, the NRC limit for this radionuclide in Class A waste is very similar to the PE intrusion limit.

The permissible concentrations estimated by the PEs for this radionuclide are within one or two orders of magnitude of those from the LLW performance assessments at Hanford, INEL, and ORR for both the water pathway and intruder concentrations. A comparison has not been made for SRS because of difficulty in interpreting the performance assessment. The PE concentrations for the water pathway are more conservative (i.e., lower) than the performance assessment results with the best comparison occurring at INEL and the poorest at ORR. The PE concentrations for the intrusion pathway match those derived on the basis of the performance assessments within an order of magnitude.

C-14

Comparisons of the permissible waste concentrations estimated in the PEs and LLW performance assessments and the NRC limits in Class A LLW for C-14 are shown in Figure 7-3(b). The permissible waste concentrations for this radionuclide are limited by the atmospheric pathway at most of the arid sites and one of the humid sites. The water pathway limits concentrations at RFETS, ANLE, FEMP, PORTS, ORR, SRS, and WVDP. Intrusion is not the limiting pathway for any site. The NRC Class A waste limit is a factor of 13 higher than the PE intruder concentrations.

The estimated waste concentration limits based on the PE analysis for the water pathway are within one or two orders of magnitude of those derived through the LLW performance assessments at Hanford, INEL, and ORR; a comparison has not been made for SRS. The PE estimates for the water pathway are more conservative (i.e., lower) than the performance assessment results, with the best comparison occurring at INEL and the poorest at ORR. The PE intrusion scenario estimates match the performance assessment estimates favorably to within an order of magnitude.

Sr-90 and Cs-137

Comparisons of the estimated permissible waste for Sr-90 and Cs-137 are shown in Figure 7-3(c) and (e). The intruder pathway limits the permissible waste concentrations for Sr-90 and Cs-137 at all sites because the concentrations based on the water pathway are unlimited. In both cases, the concentration limit based on the PE analysis for the intruder scenario agrees with the NRC Class A concentrations limit within a factor of three for these radionuclides.

The permissible concentrations for the water pathway based on the PE estimates and the performance assessments at Hanford, INEL, ORR, and SRS are not limited because the long travel times relative to the half-lives for these radionuclides cause them to decay to insignificant levels at the performance boundary. The permissible concentrations for the intruder scenarios based on performance assessments are within an order of magnitude of the PE estimates for Sr-90 at Hanford and ORR and for Cs-137 at ORR. The permissible concentrations for intrusion based on performance assessments are within two orders of magnitude for Sr-90 at INEL and for Cs-137 at Hanford and INEL.

Tc-99

Comparisons of the estimated permissible waste concentrations for Tc-99 are shown in Figure 7-3(d). The permissible concentrations for this radionuclide are limited by the water pathway at all sites except NTS, where the radionuclide arrives at the performance boundary beyond 10,000 y. In general, the concentration limits for the water pathway are more restrictive at the humid sites than at the arid sites. The NRC limit for this radionuclide in Class A LLW is about an order of magnitude higher than the PE intruder concentrations.

The permissible concentrations for the water pathway based on the performance assessments at Hanford, INEL, and SRS are within about an order of magnitude of the PE estimates. The estimate based on the performance assessment at ORR is a factor of about 50 higher than the PE estimate. In all cases where both sets of estimates are available, the PE estimates are more restrictive. The permissible concentrations for the intruder scenarios based on the performance assessments at Hanford, INEL, ORR, and SRS are within an order of magnitude of the PE estimates.

U-238

Comparisons of the estimated permissible waste concentrations for U-238 are shown in Figure 7-3(f). The permissible waste concentrations for U-238 are limited by intrusion at the arid sites (except for Hanford) and at a few humid sites (WVDP and ANLE) because the radionuclide arrives at the performance boundary beyond 10,000 y for the water pathway and is not considered in determining the permissible concentration at these sites. The water pathway limits the permissible concentrations for this radionuclide at Hanford due to the high mobility assumed at that site. The water pathway for U-238 is limiting at the remaining humid sites due to its moderate mobility and short travel times. In general, the permissible waste concentrations for this radionuclide are low at all sites. The concentration limits for the water pathway are more restrictive than the intruder limits at the humid sites by one to two orders of magnitude. No NRC Class A waste limit is specified for U-238.

The permissible concentrations for the water pathway based on the performance assessments at Hanford and INEL are within about an order of magnitude of the PE estimates. The estimate based on the performance assessment at ORR is a factor of about 50 higher than the PE estimate. In all cases where both sets of estimates are available, the PE estimates are more restrictive. An estimate of the permissible concentration for the water pathway from the performance assessment at SRS has not been made because of difficulty in interpreting the performance assessment. The permissible concentrations for the intruder scenarios based on the

performance assessments at Hanford, INEL, ORR, and SRS are within about an order of magnitude of the PE estimates.

Pu-239

Comparisons of the estimated permissible waste concentrations for Pu-239 are shown in Figure 7-3(g). In general, the permissible waste concentrations for this radionuclide are limited by intrusion because Pu-239 arrived at the performance boundary beyond 10,000 y for the water pathway at all sites but ORR. At ORR, the water pathway limited Pu-239 due to the high mobility assumed at that site, with the water pathway concentration limit being more than two orders of magnitude lower than that for intrusion. The estimated NRC limit for this radionuclide in Class A LLW is within an order of magnitude of the estimate derived through the PE intruder scenario.

The estimated permissible concentrations for the water pathway based on the performance assessments at Hanford and INEL are also based on arrival at the performance boundary beyond 10,000 y and are not shown. At ORR, the estimate based on the LLW performance assessment is a factor of about 50 higher than the PE estimate. An estimate of the permissible concentration for the water pathway from the performance assessment at SRS has not been made because of difficulty in interpreting the performance assessment. The permissible concentrations for the intruder scenarios based on the performance assessments are within about an order of magnitude of the PE estimates at all four sites.

Am-241

Comparisons of the estimated permissible concentrations for Am-241 are shown in Figure 7-3(h). The permissible waste concentrations for this radionuclide are limited by intrusion at all sites but ORR, although by only slightly at RFETS, ANLE, and PORTS. At ORR, Am-241 was limited by the water pathway due to the assumption of high mobility at that site, with the permissible waste concentration limit for the water pathway being over two orders of magnitude more restrictive than that for the intruder scenario. The estimated NRC limit for this radionuclide in Class A waste is within an order of magnitude of the PE intruder concentrations.

The permissible concentrations for the water pathway based on the performance assessments at Hanford and INEL are also based on estimated arrival at the performance boundary beyond 10,000 y and are not shown. At SRS, the estimate based on the LLW performance assessment is a factor of 2 higher than the PE estimate. At ORR, the estimate based on the LLW performance assessment is a factor of about 50 higher that the PE estimate. The permissible concentrations for the intruder scenarios based on the performance assessments are within about an order of magnitude of the PE estimates at Hanford, INEL, and ORR, while the estimate based on the performance assessment at SRS was approximately two orders of magnitude higher than the PE estimate.

7.5.2 Individual Radionuclides

The indicator radionuclides are useful for discussing the basis for the trends in limiting pathways at the 15 sites. With this basis, a summary of the limiting pathways for all 58 radionuclides at the 15 sites is now presented. The number of the 58 radionuclides controlled by the water and atmospheric pathways and intrusion scenarios for the 15 sites is shown in Figure 7-4. This figure confirms the previous observation that the permissible waste concentrations for disposal of most radionuclides at arid sites are predominantly limited by intrusion scenarios. In contrast, the water pathway is limiting for a significant number of radionuclides at most humid sites, the exceptions being ANLE and WVDP.

The radionuclides limited by the water and atmospheric pathways for both the generic trench and tumulus are shown in Table 7-10 for the 15 sites. All blank cells and radionuclides not listed are limited by the intrusion pathway. Analysis of this table shows the following:

Tritium (H-3) is limited by the water pathway at Hanford, RFETS, and the humid sites (except WVDP) for the generic trench and at ORR and SRS for both the generic trench and tumulus. At all other sites the generic tumulus detains this radionuclide for a sufficient period of time to make the limiting permissible waste concentration that based on evaluations of the intruder scenarios.

For both the generic trench and tumulus designs, the atmospheric pathway is limiting for C-14 at all arid sites except RFETS; the water pathway is limiting for C-14 at all humid sites except PGDP.

The water pathway is limiting for Tc-99 at all sites but one for both the generic trench and tumulus designs. The exception is NTS, which is assumed to have no water pathway.

Several radionuclides (Cl-36, K-40, Pd-107, I-129, U-233, U-234, U-235, U-236, U-238, and Np-237) are limited by the water pathway at most humid sites for both the generic trench and tumulus designs. These radionuclides are long-lived and relatively mobile in the environment. Several other radionuclides are limited by the water pathway at selected sites.

Fourteen radionuclides not listed in Table 7-10 (Al-26, Co-60, Ni-63, Sr-90, Ag-108m, Cd-113m, Sn-121m, Ba-133, Cs-137, Sm-151, Eu-152, Eu-154, Pb-210, and Ra-228) are limited by intrusion at all sites. These results are due to either (1) high radionuclide K_d values in soil that cause arrival at the performance boundary beyond10,000 y or (2) a combination of short half-life and medium to high radionuclide K_d values in soil that cause significant decay prior to reaching the performance boundary. This being the case, disposal of these radionuclides is possible at all 15 sites with the same permissible waste concentration.

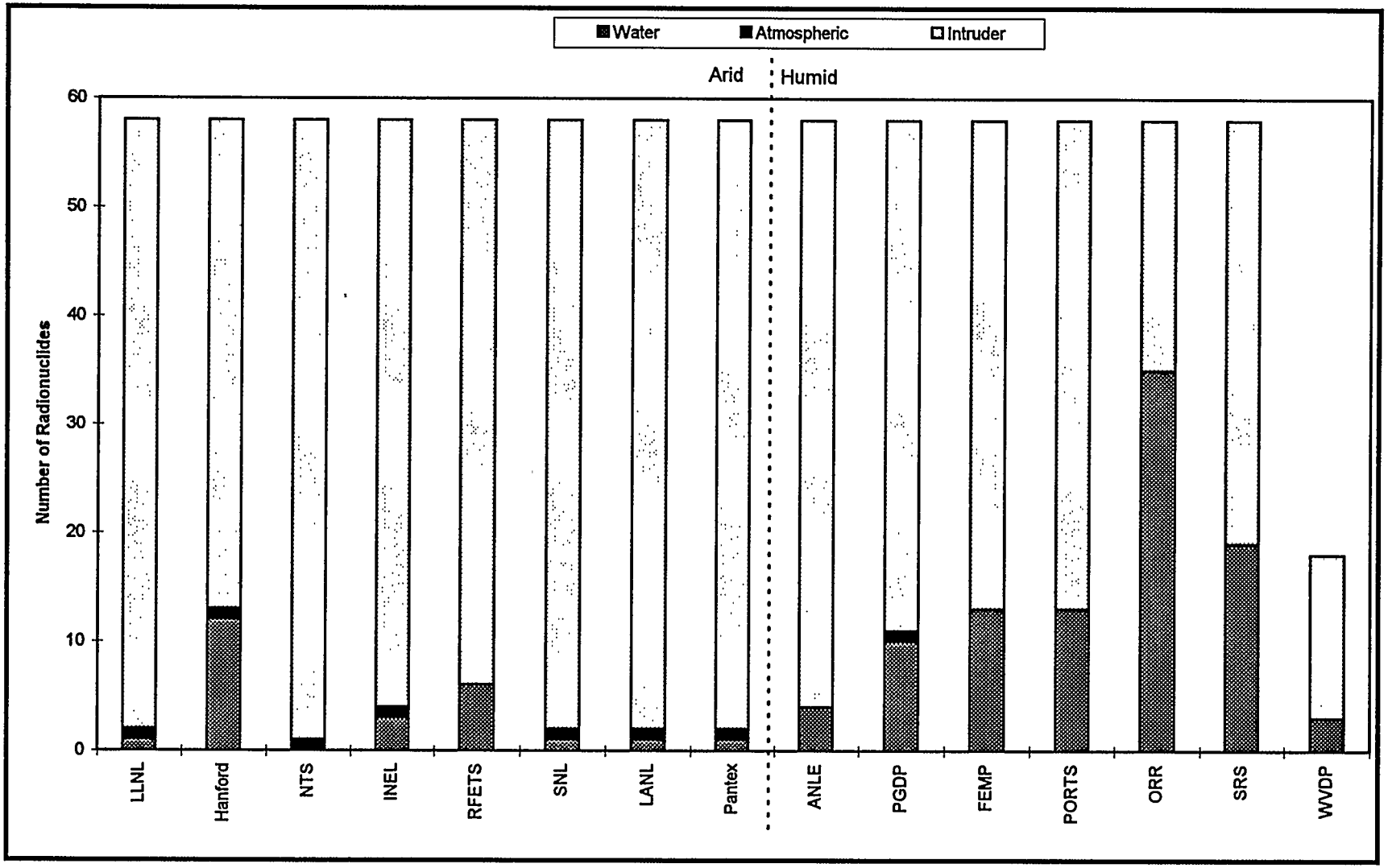


Figure 7-4. Number of radionuclides limited by the water and atmospheric pathways and intrusion scenarios (tumulus design).

Table 7-10. Radionuclides Limited by the Water Pathway for the Generic Trench Only (o), for Both the Generic Trench and Tumulus (•), and for the Atmospheric Pathway for Both Facility Types (x)

Nuclide[a]	Arid Sites								Humid Sites							Nuclide
	LLNL	Hanford	NTS[b]	INEL	RFETS	SNL	LANL	Pantex	ANLE	PGDP	FEMP	PORTS	ORR	SRS	WVDP[c]	
H-3		o			o				o	o	o	o	•	•		H-3
C-14	x	x	x	x	•	x	x	x	•	x[d]	•	•	•	•	•	C-14
Si-32														•	-[e]	Si-32
Cl-36		•			•				o	•	•	•	•	•	-	Cl-36
K-40				•	•				o	•	•	•	•	•		K-40
Ni-59												o			-	Ni-59
Se-79		•									o	o	•	•	-	Se-79
Zr-93														•	-	Zr-93
Nb-93m														•	-	Nb-93m
Nb-94													o		-	Nb-94
Tc-99	•	•		•	•	•	•	•	•	•	•	•	•	•	•	Tc-99
Pd-107										o	•	•	•	•	-	Pd-107
Sn-126												o	•		-	Sn-126
I-129		•		•	•				•	•	•	•	•	•	•	I-129
Cs-135		•										o		o	-	Cs-135
Ra-226												o			-	Ra-226
Th-229													•		-	Th-229
Th-230													•		-	Th-230
Th-232													•		-	Th-232
Pa-231		•												•	-	Pa-231
U-232		•									•					U-232
U-233		•								•	•	•	•	•		U-233
U-234		•								•	•	•	•	•		U-234
U-235		•								•	•	•	•	•		U-235
U-236		•								•	•	•	•	•		U-236
U-238		•								•	•	•	•	•		U-238
Np-237					•				•	•	•	•	•	•	-	Np-237
Pu-238												•	•	•		Pu-238
Pu-239													•	o		Pu-239
Pu-240													•	o		Pu-240
Pu-241					o				o				•	o	-	Pu-241
Pu-242													•	o	-	Pu-242
Pu-244													•	o	-	Pu-244
Am-241					o				o			o	•			Am-241
Am-243													•		-	Am-243
Cm-243													•	o	-	Cm-243
Cm-244													•	o	-	Cm-244
Cm-245													•		-	Cm-245
Cm-246													•		-	Cm-246
Cm-247													•		-	Cm-247
Cm-248													•		-	Cm-248
Cf-249													•		-	Cf-249
Cf-250													•		-	Cf-250
Cf-251													•		-	Cf-251

a Fourteen radionuclides not listed--all intruder limited (Al-26, Co-60, Ni-63, Sr-90, Ag-108m, Cd-113m, Sn-121m, Ba-133, Cs-137, Sm-151, Eu-152, Eu-154, Pb-210, and Ra-228).

b No water pathway analysis was performed at this site.

c Only 18 on-site radionuclides were evaluated.

d Trench is limited by the water pathway.

e "-" indicates radionuclide not evaluated at this site

An additional seventeen of the 58 radionuclides (Si-32, Ni-59, Zr-93, Nb-93m, Nb-94, Ra-226, Th-229, Th-230, Th-232, Am-243, Cm-245, Cm-246, Cm-247, Cm-248, Cf-249, Cf-250, and Cf-251) are limited by intrusion at 14 of the 15 sites. These results are due to either (1) high radionuclide K_d values in soil that cause arrival at the performance boundary beyond 10,000 y or (2) a combination of short half-life and medium to high radionuclide K_d values in soil that cause significant decay prior to reaching the performance boundary. The water pathway limits Si-32, Zr-93, and Nb-93m at SRS; Ni-59, and Ra-226 at PORTS; and Nb-94, Am-243, and the radioisotopes of thorium, curium, and californium at ORR. Based on the PE results, the limiting concentrations for disposal of these 17 radionuclides at 14 of the 15 sites are those based on the intruder pathway.

An additional 9 of the 58 radionuclides (Sn-126, Pa-231, U-232, Pu-239, Pu-240, Pu-242, Pu-244, Cm-243, and Cm-244) are limited by intrusion at 13 of the 15 sites. The water pathway limits Sn-126 at PORTS and ORR; Pa-231 at Hanford and SRS; U-232 at Hanford and FEMP; and the radioisotopes of plutonium and curium at ORR and SRS. These results are due to either (1) high radionuclide K_d values in soil that cause arrival at the performance boundary beyond 10,000 y or (2) a combination of short half-life and medium to high radionuclide K_d values in soil that cause significant decay prior to reaching the performance boundary. Thus, the PE results indicate that in the case of these 9 radionuclides, the permissible waste concentrations are based on the intruder scenarios at 13 of the 15 sites.

For the humid sites, a larger number of radionuclides are limited by the generic trench design than by the tumulus design. This indicates that the tumulus design is more effective than the trench design at such sites. However, at some of these sites the shallow depth of the groundwater precludes subsurface (i.e., trench) disposal.

At NTS, 57 of the 58 radionuclides are limited by the intrusion pathway, and C-14 is limited by the atmospheric pathway; the water pathway was not evaluated. The NTS has the highest overall permissible waste concentrations (based on technical considerations) of the 15 sites evaluated.

At LLNL, SNL, LANL, and Pantex, 56 of the 58 radionuclides are limited by intrusion, C-14 is limited by the atmospheric pathway, and Tc-99 is limited by the water pathway. The differences in disposal performance at these four sites are almost indistinguishable using the PE methodology. The permissible waste concentrations are only slightly lower than at NTS where the water pathway is virtually non-existent.

At Hanford, all the uranium isotopes are limited by the water pathway due to its assumed high environmental mobility at that site.

At ORR, 35 of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus designs and one radionuclide is limited by the water pathway for the generic trench design. The ORR site has the most restrictive permissible waste concentrations of the 15 sites.

At SRS, 19 of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus and an additional 8 radionuclides are limited by the water pathway for the generic trench. Thirteen of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus at FEMP and PORTS, and an additional 2 and 7 radionuclides are limited by the water pathway for the generic trench at FEMP and PORTS, respectively. At PGDP, 10 of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus and an additional 3 radionuclides are limited by the water pathway for the generic trench. These results provide an indication of the increased effectiveness of a tumulus facility at these humid sites. Of course, at some of these sites, the shallow depth of groundwater precludes subsurface (i.e., trench) disposal.

7.6 DISCUSSION

The effects of the major assumptions in the conceptual models used in the PE and the parameter sensitivity analyses for the models are discussed in Chapter 6 of this volume. Highlights of that chapter are summarized below.

7.6.1 Conceptual Model Assumptions

As shown in Section 7.5.1, many of the assumptions used in the PE result in permissible waste concentrations at the various sites that, in general, are more restrictive (lower) than those produced through the detailed performance assessment analyses. The one exception is INEL where the performance assessment limit for H-3 in the case of the intruder scenario is an order of magnitude more restrictive than the PE limit. However, despite this conservatism, as shown in Figure 7-4 the majority of the 58 radionuclides at all sites except ORR are limited by the assumed intrusion scenarios and not the water pathway. This result indicates that, in general, more detailed water pathway analyses might not be necessary for many radionuclides at most sites because the intruder scenario results are more limiting. However, results from intruder scenarios should be evaluated carefully because future social behavior, and intrusion scenarios, are resistant to prediction.

7.6.1.1 All Pathways and Scenarios

Major assumptions used in the PE that affect estimates of the permissible waste concentrations based on analyses of the water pathway, the atmospheric pathway, and the intruder scenarios include (1) the anticipated performance of the waste form and (2) the anticipated performance of the engineered barriers.

Although the PE considered a grouted waste form, performance of other waste forms can be easily substituted into the PE methodology provided representative data describing the performance of the specified waste form are available (see Chapter 6 of Volume 2). Vitrified waste and debris waste represent two potential alternative waste forms. Near surface disposal of a vitrified waste form would likely result in permissible waste concentrations, based on water pathway analyses, that are several orders of magnitude higher than those resulting from grouted wastes. These higher concentrations are due to the much smaller quantities of radionuclides anticipated to be present in the leachate from vitrified waste. In addition, the process of vitrifying

MLLW would likely volatilize several radionuclides (e.g., H-3, C-14). Although this process may present an additional health risk to workers, the intrusion scenarios might have fewer health effects because of greater difficulty intruding into glass than into concrete as well as differences in particle sizes likely to be generated.

In terms of the water pathway and the intruder scenarios, assumptions about the performance of engineered barriers primarily affect the short-lived radionuclides. Because such radionuclides are generally limited by the post-drilling intrusion scenario, which is assumed to occur at 100 y after disposal (the assumed period of institutional controls), the permissible waste concentrations for shorter-lived radionuclides would increase if longer-duration engineered barriers could be justified.

Performance of engineered barriers is difficult to justify for more than a few hundred to a few thousand years for several reasons: the long-term performance of concrete or grout is not well known and the dilution credit for grouted wastes will likely change; vault technologies may be more or less robust; and the performance of the leachate collection/drainage systems is largely ignored. Because little is known about the long-term performance of the waste form and disposal technology, relatively short time periods are used in the analyses of these features.

7.6.1.2 Water Pathway

The site-specific conceptual models and associated parameters for the PE water pathway analysis were developed through close interactions with the site technical staff. The models and parameters were subsequently reviewed by the project's Internal Review Team and Senior Review Panel. The PE analyses are simplified representations of site behavior using a number of conservative assumptions relative to the transport of the radionuclides. The PE methodology assumes that the peak concentrations of radionuclides in the groundwater can only be reduced by dilution of the leachate by the groundwater and through radioactive decay. The major assumptions used in the water pathway analysis relate to: (1) solubility constraints, (2) continuous source, (3) sorption effects, (4) fractured flow, and (5) dilution due to regional recharge. The effects of each are summarized below.

The highest aqueous concentrations of radionuclides are in the leachate exiting the disposal facility (determined by the ratio of radionuclide concentration in the grout to the CRF_{Source} [which is related to the grout K_d]). Leachate concentrations may be limited by the solubility of the radionuclides—a mechanism not considered in the PE analysis. Incorporating the effects of solubility into the analysis might increase some permissible radionuclide concentrations in the waste relative to those estimated in the PE. These increases could occur because the solubility constraint would prevent increases in leachate concentrations proportional to radionuclide concentrations in waste.

The highest radionuclide concentrations in leachate from among the 15 sites were compared with estimates of solubility limits for the indicator radionuclides. For about half of the indicator radionuclides (Cs-137, U-238, and Pu-239), the solubility limit was lower than the leachate concentration assumed in the PE (see Table 6-2). The concentrations for Cs-137 is unlimited for the water pathway at most sites, so application of solubility limits does not provide

higher limits. For U-238 and Pu-239, the solubility limit is about 50 and 5 times lower, respectively, than the leachate concentration, indicating that, for the water pathway, consideration of solubility limits would increase the permissible waste concentrations for these radionuclides and is therefore conservative. Because solubility limits are dependent on the chemical composition of the leachate and groundwater, especially for uranium and plutonium, the values used elsewhere may be different than those discussed here.

The continuous-source assumption provides minor conservatism when compared to a finite-duration source because it allows dispersion effects to be ignored. This assumption has also been used because the radionuclide inventories needed for estimating the duration of release from the disposal facility at each site are not known. Compared with the LLW performance assessment at INEL that included dispersion but otherwise used an analysis similar to the PE, the PE permissible waste concentrations are about 15% lower (more conservative). Effects may be different at other sites.

Sorption is treated with a linear and reversible equilibrium mechanism in the PE. This treatment results in retardation of the radionuclides but no concentration attenuation except by decay because a continuous source is assumed. The linear and reversible equilibrium mechanism is a common way to treat sorption in performance assessments, although other mechanisms have been used (e.g., partially reversible sorption, kinetically limited sorption). The approach use in the PE provides conservative results relative to these other sorption models.

Fracture flow and flow through preferential flow paths are treated conservatively by (1) assuming that the vadose zone in the PE is equal to the non-fractured portion or non-preferential flow path portion only and (2) combining the fractured porosity with the matrix porosity as an effective porous media in the saturated zone. The vadose zone method is consistent with the approach used in the LLW performance assessment at INEL. The saturated zone method is commonly used in performance assessments. Fractured flow was assumed to occur at LANL, ORR, and ANLE.

While dilution due to mixing of leachate with uncontaminated groundwater is included in the analysis, dilution of radionuclide concentrations in the saturated zone by regional recharge is not considered in the PE. Dilution of contaminants in the saturated zone due to regional recharge can increase the permissible waste concentrations slightly. Ignoring this mechanism represents a conservatism in the PE. Humid sites with high recharge relative to groundwater flow and low CRF_{Water} values would benefit most from dilution from regional recharge, although the effect would be minor.

7.6.1.3 Atmospheric Pathway

Two major assumptions are used to develop the conceptual model of the atmospheric pathway: (1) vapor diffusion is used as a surrogate for other transport processes and (2) the radionuclides for which analyses were performed were restricted to those with a volatility and chemical form that made them candidates for airborne release.

Of the many potential processes for transporting radionuclides from a disposal facility to the soil surface (e.g., gas and vapor diffusion, desiccation cracks, erosion, plant root uptake, and burrowing animals), only vapor diffusion is used in the PE to evaluate the atmospheric releases. Conservative parameter values have been used in the PE models for gas and vapor diffusion to attempt to capture the uncertainty of representing other transport mechanisms.

Only two radionuclides (H-3 and C-14) were evaluated in terms of the atmospheric pathway. One hundred years of detention in the disposal facility is assumed prior to their release. The radionuclide H-3 must become associated with water vapor and migrate from the hygroscopic Portland cement-based grouted waste, and C-14 must be transformed into an acid gas (CO_2) (a form unlikely to exist in the near future in the high pH environment associated with grouted waste) prior to its migration from the disposal facility. The assumed RCRA-compliant liner and cover systems also present barriers to migration of the volatile radionuclides. Therefore, the 100-y time period used in the analysis may be reasonable. Because it is short-lived, H-3 will decay to insignificant levels after a period of 100 y or more. In contrast, because C-14 is long-lived, detention times in this range, and even longer, will have little effect on the permissible concentrations for this radionuclide.

7.6.1.4 Intruder Scenarios

Two major assumptions were made in the conceptual model used for estimating the permissible waste concentrations based on intrusion: (1) the identification and selection of the scenarios assumed to be applicable; and (2) the time at which intrusion is assumed to occur. The selection of applicable scenarios is largely a matter of policy. However, the scenarios selected for this analysis are based on those used by NRC (1982) and by DOE (ORNL, 1994 and MMES et al., 1994).

For the purposes of the PE, the homesteader and post-drilling scenarios are assumed to occur at all sites. However, it is recognized that, under some conditions, the homesteader scenario may not be credible. For example, when disposal of the waste is below grade at a depth sufficient to preclude intrusion for the construction of a basement, this particular scenario is unrealistic. For these situations, the post-drilling intrusion scenario is used to estimate the permissible waste concentrations for intrusion. The net effect is that the permissible waste concentrations for disposal of radionuclides in the generic trench and tumulus designs are increased for 47 and 43 of the 58 radionuclides, respectively. For the medium- and longer-lived radionuclides, these increases range up to three orders of magnitude; the permissible concentrations for the shorter-lived radionuclides are already limited by the post-drilling scenario.

The time of intrusion for the homesteader scenarios is 300 y and 500 y for the generic trench and tumulus designs, respectively, while it is generally 100 y for the post-drilling scenario for both facility designs. Assuming that the homesteader scenario occurs at 100 y instead of 300 y and 500 y for the generic trench and tumulus facilities, respectively, causes the shorter-lived radionuclides to have reduced permissible waste concentrations by up to three orders of magnitude relative to the post-drilling scenario, which, as previously stated, generally limits the permissible concentration for the shorter-lived radionuclides. The permissible concentrations for the longer-lived radionuclides are minimally affected by this change of intrusion times.

7.6.2 Parameter Sensitivity Analyses

A discussion of the parameter sensitivity analysis for the water and atmospheric pathways and intrusion scenarios is presented in Chapter 6 of this volume. The effects of changes in the values for the most sensitive parameters on the permissible concentrations are further analyzed in each chapter in Volume 3. The overall results of these analyses can be summarized as follows.

Changes to five parameters in the water pathway were shown in Chapter 6 to provide potentially significant changes in the estimated permissible concentrations: grout distribution coefficients, K_d^G; annual recharge at a site, i; groundwater Darcy velocity at a site, q_{gw}; plan area of the disposal facility, A; and soil distribution coefficients, K_d. The results were relatively insensitive to the remaining parameters due to either (1) limitations on the physical range of the parameter (e.g., bulk density and porosity) or (2) a high degree of certainty in the measured value (e.g., thickness of the vadose zone).

The radionuclide-specific K_d^G values are assumed to be the same at all sites so that changes in this parameter will cause identical changes at all sites. The site-specific impacts of changes in the remaining four parameters are presented in the site chapters with respect to changing limiting permissible waste concentrations from/to the water pathway to/from the intrusion scenario or atmospheric pathway. Sensitivity analyses showed that, for all sites, large variations in any single parameters were required to change the pathway that was limiting in terms of permissible waste concentrations.

The two most important parameters in the water pathway calculations are the natural recharge and the K_d values of radionuclides in the subsurface media; both parameters affect the travel time of radionuclides. Because of the 10,000-y limiting performance period used in the PE analysis, K_d values used for those radionuclides that do not arrive at the performance boundary until after 10,000 y can be very important. For example, U-238 and Pu-239 do not reach the performance boundary in the groundwater at SNL before 10,000 y. Sensitivity analysis shows that the K_d values of U-238 and Pu-239 in the subsurface soils must be lowered from 35 mL/g and 550 mL/g, respectively, to 0.75 mL/g to decrease the travel time to less than 10,000 y. Another example is Tc-99 at LLNL, where the water pathway is the controlling pathway when an recharge of 0.025 m/y is assumed. Sensitivity analysis shows that the natural recharge would have to decrease to 0.0018 m/y, more than an order of magnitude, for the intrusion pathway to control the permissible concentration for this radionuclide.

The atmospheric pathway was designed to provide estimates of permissible concentrations as a surrogate for a variety of mechanisms (e.g., burrowing animals, bio-intrusion, and desiccation cracks), and many of the parameters used in the analysis are based on literature values. Due to the generic nature of the analysis, the sensitivity analysis shows that no reasonable site-specific parameter changes would significantly change the results.

A generic, mostly site-independent approach to intrusion analysis has been adopted in the PE analysis. Consequently, on the basis of these analyses, the permissible waste concentration for each radionuclide is the same at all sites regardless of site-specific conditions except at SRS. At

that site, the post-drilling intrusion scenario was modified to reflect local drilling conditions. The sensitivity analysis for the intrusion scenarios provided information on the most significant exposure pathways (e.g., inhalation; ingestion of water, agricultural products, and soil; external exposure) for the various radionuclides. Increasing the values of the parameters to their reasonable upper bounds based on physical limitations or constraints resulted in changes to the permissible radionuclide concentrations in waste of factors less than or equal to five for the indicator radionuclides. The largest changes to the permissible radionuclide concentrations in waste resulted from changes in the mixing fraction (the ratio of exhumed waste mixed with clean soil).

7.6.3 Comparison of the PE with LLW Performance Assessments

Appendix A of Volume 3 compares the results of the PE analyses to the results from site-specific performance assessments at INEL, Hanford, ORR, and SRS. While the performance assessments attempt to be conservative representations of actual site behavior, the PE water pathway analyses provide more conservative (i.e., lower) permissible waste concentrations than the performance assessments due to the simple and conservative transport assumptions used in the PE. For example, non-equilibrium sorption models were used in the performance assessments at ORR and Hanford for the transport in the vadose zone. Because using these models leads to additional attenuation of the peak radionuclide concentrations that are not reflected in the PE calculations, the estimates associated with the PE tend to be more conservative.

A summary of the comparison of the permissible waste concentrations in the PE and LLW performance assessments is shown in Table 7-11 for radionuclides arriving at the performance boundary before 10,000 y along with the differences in transport mechanisms used in the two analyses. The PE estimated concentrations for the water pathway are within two orders-of-magnitude of the performance assessment results for all radionuclides, and less than one order of magnitude for most radionuclides, at a few sites despite the differences in the methodologies. The use of multiple transport mechanisms in the performance assessments generally resulted in the largest differences between the PE and performance assessment results.

As demonstrated in Appendix A of Volume 3, the PE analysts were able to reproduce the performance assessment results when the performance assessment assumptions were incorporated into the PE analyses. This agreement between the PE and performance assessments results should not be viewed as a validation of the PE or the performance assessment analyses. The PE conceptual models were based largely on the performance assessment conceptual models at these sites, and many of the same parameters were used in both analyses. The PE analyses can be viewed as a conceptual and mathematical simplification of the more sophisticated and complicated performance assessment computer codes; similar results can be reproduced with simple analyses using the same conceptual models.

Table 7-11. Comparison of the Permissible Waste Concentrations and Assumptions in the PEs and LLW Performance Assessments for the Water Pathway

Performance Assessment Site	Ratio of Value from Performance Assessment to PE Value for C_W	Additional Transport Mechanisms Used in the Performance Assessment
Hanford	10-20	• Non-equilibrium sorption used in the vadose zone. • Longitudinal dispersion used in the vadose and saturated zones. • Dilution due to mixing with ambient water in the vadose zone.
INEL	1-1.2	• Longitudinal dispersion used in the vadose and saturated zones. • Dose from decay products not considered.
ORR	20-50	• Non-equilibrium sorption used in the vadose zone. • Dilution due to mixing with regional recharge water in the saturated zone. • Longitudinal dispersion used in the saturated zone.
SRS	1-10	• Constant source due to solubility constraints for uranium and plutonium.

The intruder analyses used in the PE are generally based on the methods presented in the ORR and SRS performance assessments. As shown in Appendix A of Volume 3, the PE waste concentration limits are similar to those derived on the basis of performance assessments for these two sites. Additionally, the PE estimated concentrations are generally similar to those derived from the performance assessments for the intruder scenarios at INEL and Hanford.

7.7 OBSERVATIONS

Many factors are important in developing and comparing MLLW disposal options, including selection of waste treatment and stabilized waste form, disposal facility design, distributive equity, transportation risks and costs, and social and political factors. As applied here, the PE is a simple, scoping-level analysis which primarily provides technical information on the relative capability of 15 DOE sites to dispose of 58 radionuclides in generic trench and tumulus facilities. Additionally, these facilities are assumed to satisfy the relevant design requirements of RCRA for the hazardous constituents of the MLLW.

The observations based on the analyses contained in this report are grouped into four categories: (1) site performance, (2) radionuclide performance, (3) sensitivity of results, and (4) comparisons with other relevant analyses. The 15 sites analyzed in this report are classified as "arid" or "humid" according to their climatological characteristics. The sites classified as arid are LLNL, Hanford, NTS, INEL, RFETS, SNL, LANL, and Pantex. The sites classified as humid are ANLE, PGDP, FEMP, PORTS, ORR, SRS, and WVDP.

Site Performance

- At the arid sites, the permissible waste concentrations for most radionuclides are limited by the intruder pathway because the travel times to the performance boundary for the water pathway are generally sufficient to allow (1) significant radioactive decay for the short-lived radionuclides and (2) arrival beyond the performance period of 10,000 y for all but the long-lived, highly mobile radionuclides.

- At most humid sites, the permissible waste concentrations for many radionuclides are limited by the intruder pathway because the travel times to the performance boundary for the water pathway are generally sufficient to allow (1) significant radioactive decay for the less-mobile, short-lived radionuclides and (2) arrival beyond the performance period of 10,000 y for all but the few long-lived, mobile radionuclides.

- At NTS, 57 of the 58 radionuclides are limited by the intrusion pathway, and C-14 is limited by the atmospheric pathway. The water pathway was not analyzed at this site because it was not considered a credible pathway due to the extremely arid nature of the site and the large depth to groundwater. The NTS has the highest overall permissible waste concentrations (based on technical considerations) of the 15 sites evaluated.

- At LLNL, SNL, LANL, and Pantex, 56 of the 58 radionuclides are limited by intrusion, C-14 is limited by the atmospheric pathway, and Tc-99 is limited by the water pathway. The differences in disposal performance at these four sites are almost indistinguishable using the PE methodology. The permissible waste concentrations are only slightly lower than at NTS where the water pathway is virtually non-existent.

- At Hanford, the PE results suggest that all the uranium isotopes are limited by the water pathway due to the observed high mobility of these isotopes at that site. However, uranium generally has low solubility, and including consideration of solubility limits may change this result.

- At ORR, 35 of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus designs, and one radionuclide is limited by the water pathway for the generic trench design. Based on the analysis in the PE, ORR has the most restrictive permissible waste concentrations of the 15 sites for radioactive waste disposal.

- At SRS, 19 of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus designs, and an additional 8 radionuclides are limited by the water pathway for the generic trench design. The remaining radionuclides were limited by intrusion.

- At FEMP and PORTS, 13 of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus designs; an additional 2 and 7 radionuclides are limited by the water pathway for the generic trench design at FEMP and PORTS, respectively. At PGDP, 10 of the 58 radionuclides are limited by the water pathway for both the generic trench and tumulus designs, and an additional 3 radionuclides are limited by the water pathway for the

generic trench design. The remaining radionuclides were limited by intrusion except C-14 in the generic tumulus at PGDP, where this radionuclide is limited by the atmospheric pathway.

- At the humid sites, more radionuclides are limited by the water pathway than at the arid sites. This suggests that the longer containment provided by the tumulus design is of special benefit at these locations because fewer radionuclides are limited by this pathway than for the trench design. It should be noted, also, that at some of these sites the shallow depth of the groundwater precludes subsurface (i.e., trench) disposal.

Radionuclide-Specific Performance

- Tritium (H-3) is limited by the water pathway at Hanford, RFETS, and the humid sites (except WVDP) for the generic trench design and at ORR and SRS for the both the generic trench and tumulus designs. At most sites, the generic tumulus design provides sufficient detention for disposal of H-3 at the intruder concentration limit. Tritium is not limited by the atmospheric pathway at any of the sites.

- For both the generic trench and tumulus designs, the atmospheric pathway is limiting for C-14 at all arid sites except RFETS; the water pathway is limiting for C-14 at all humid sites except PDGP. The limiting concentrations for C-14 based on the water pathway or atmospheric pathway are generally within an order of magnitude at all sites.

- The water pathway is limiting for Tc-99 at all sites but one for both the generic trench and tumulus designs. The exception is NTS, which is assumed to have no water pathway. This is due to its high mobility and long half-life. In general, the permissible waste concentrations for this radionuclide are moderately higher at the arid sites than at the humid sites; at NTS, the permissible concentration is significantly higher.

- Several radionuclides (Cl-36, K-40, Pd-107, I-129, U-233, U-234, U-235, U-236, U-238, and Np-237) are limited by the water pathway at most humid sites for both the generic trench and tumulus designs. These radionuclides are long-lived and relatively mobile in the environment.

- Fourteen of the 58 radionuclides (Al-26, Co-60, Ni-63, Sr-90, Ag-108m, Cd-113m, Sn-121m, Ba-133, Cs-137, Sm-151, Eu-152, Eu-154, Pb-210, and Ra-228) are limited by intrusion at all sites. These results are due to either arrival at the performance boundary beyond 10,000 y or significant decay prior to reaching the performance boundary. This being the case, disposal of these radionuclides is possible at all 15 sites with the same permissible waste concentration.

- An additional seventeen of the 58 radionuclides (Si-32, Ni-59, Zr-93, Nb-93m, Nb-94, Ra-226. Th-229, Th-230, Th-232, Am-243, Cm-245, Cm-246, Cm-247, Cm-248, Cf-249, Cf-250, and Cf-251) are limited by intrusion at 14 of the 15 sites. The water pathway limits Si-32, Zr-93, and Nb-93m at SRS; Ni-59 and Ra-226 at PORTS; and Nb-94, Am-243, and the radioisotopes of thorium, curium, and californium at ORR. These results are due to either arrival at the performance boundary beyond 10,000 y or significant decay prior to reaching the performance boundary. Based on the PE results, the limiting concentrations for disposal of these 17 radionuclides at 14 of the 15 sites are those based on the intruder scenarios.

- An additional ten of the 58 radionuclides (Sn-126, Pa-231, U-232, Pu-239, Pu-240, Pu-242, Pu-244, Cm-243, and Cm-244) are limited by intrusion at 13 of the 15 sites. The water pathway limits Sn-126 at PORTS and ORR; Pa-231 at Hanford and SRS; U-232 at Hanford and FEMP; and the radioisotopes of plutonium and curium at ORR and SRS. These results are due to either arrival at the performance boundary beyond 10,000 y or significant decay prior to reaching the performance boundary. Thus, the PE results indicate that, in the case of these 10 radionuclides, the permissible concentrations are based on the intruder scenarios.

- In the case of the intrusion-limited radionuclides, the post-drilling scenario provides more restrictive limits than the homesteader scenario for those with half-lives less than about 30 y for the generic trench and less than about 130 y for the generic tumulus. This result is due primarily to the differences in times of intrusion and number of exposure pathways for the two scenarios: the post-drilling scenario occurs earlier but has fewer exposure pathways than the homesteader scenario.

Sensitivity of Results

- For radionuclides limited by the water pathway, the permissible concentrations are most sensitive to the assumed values of the grout distribution coefficient, K_d^G, at all sites. This parameter has a controlling effect on the radionuclide concentration in the leachate exiting the disposal facility. However, generic values were used for the grout distribution coefficient in the PE, so variations in the coefficient affect all sites in the same manner.

- Consideration of radionuclide solubility in the analysis may increase the permissible radionuclide concentrations for some radionuclides limited by the water pathway, particularly for the isotopes of uranium and plutonium that generally have low solubility. Radionuclides that are solubility limited will have no inventory limit.

- The travel time to the performance boundary is principally determined by the recharge and depth to groundwater for the water pathway. While depth to groundwater is generally well known at each site, average recharge must generally be estimated. The sensitivity analyses have shown that for these sites, the recharge would usually have to change by an order of magnitude or more to alter the limiting pathway for most radionuclides. The humid sites generally require smaller changes in recharge than the arid sites to change the limiting pathway.

- For the shorter-lived radionuclides at the humid sites, their permissible concentrations for the water pathway are sensitive to values of their soil distribution coefficient, K_d, and the natural recharge, i, for the site. Large changes in these parameters, however, are typically required to change the limiting pathway for most radionuclides.

- Assuming that the homesteader intrusion scenario occurs at 100 y, the same time as the post-drilling scenario, instead of at later times, causes a decrease in permissible concentration of up to three orders of magnitude for some shorter-lived radionuclides that would otherwise be limited by the post-drilling scenario. However, this assumption is not reasonable for disposal systems constructed with engineered barriers that would last beyond the 100-y period of

active institutional controls. This assumption will require further clarification in site-specific performance assessments.

- Eliminating homesteader intrusion as a credible scenario (e.g., by disposing waste below grade at a depth sufficient to preclude intrusion) results in the permissible radionuclide concentrations being determined solely by the post-drilling scenario. The permissible concentrations would be increased for 47 and 43 of the 58 radionuclides for the generic trench and tumulus designs, respectively. The permissible concentrations would be increased by up to three orders of magnitude, primarily for the medium- and longer-lived radionuclides, because the shorter-lived radionuclides are already limited by the post-drilling scenario.

- The permissible concentrations for the radionuclides released through the atmospheric pathway are not sensitive to changes in site-specific parameters because, although site-specific parameters are used, the analyses are basically generic.

- Even with the simple and conservative water pathway analyses used in the PE, many radionuclides appear to be intruder limited across the complex, indicating that a detailed water pathway analysis may not need to be considered at many sites unless important site-specific details have not been captured in the PE analysis, or the site is expected to receive large amounts of long-lived and mobile radionuclides. However, results based on intruder scenarios should be evaluated carefully because future social behavior, and intrusion scenarios, are difficult to predict.

Comparisons with Other Relevant Analyses

- The permissible radionuclide concentrations calculated for the intrusion scenarios in the PE are generally within an order of magnitude of the limits specified by the NRC in 10 CFR Part 61 for commercial disposal of Class A LLW. However, the water pathway produced permissible concentrations that are much lower than these limits for some highly mobile radionuclides (e.g., Tc-99).

- In general, the results of the PE compare reasonably well with those of the available site-specific LLW performance assessments; however, differences in permissible concentrations for specific radionuclides greater than two orders of magnitude have been found. For the water pathway, the PE analyses yield more conservative (i.e., lower) permissible waste concentrations than the performance assessments. For the intrusion scenarios, the PE and performance assessment results generally agree within an order of magnitude. In the cases in which agreement was not as good, the lack of agreement was attributable to differences in assumptions between the PE and performance assessment methodologies for a particular site.

7.8 CONCLUSIONS

- All 15 DOE sites considered in this analysis have the technical capability for disposal of some radioactive materials in mixed low-level waste. This conclusion is based on the concentration limits that were estimated using the pathways for release of radionuclides to water and the atmosphere and the assumed scenarios for inadvertent human intrusion into disposal facilities.

However, the technical capabilities for disposal of radioactive materials in mixed waste also appear to differ significantly among the sites. Differences of up to four orders of magnitude in the estimated concentration limits have been calculated for some radionuclides at the various sites when the limits are based on the most restrictive of the results for the water and atmospheric pathways and intrusion scenarios at each site. For some radionuclides, even greater differences were seen in the separate results for the water release pathway among the various sites, due primarily to the differences in the assumed water travel times between humid and arid sites. The inadvertent human intrusion scenarios used in the analysis were largely generic and did not distinguish between sites.

- For most radionuclides, the assumed scenarios for inadvertent human intrusion were more important in determining the estimated concentration limits for disposal than the scenarios for release to water or the atmosphere, particularly for sites located in arid regions. The intrusion scenarios considered in this analysis were based on scenarios commonly used in performance assessments for DOE facilities disposing of low-level radioactive waste. The scenarios are largely generic and, thus, the estimated radionuclide concentration limits are the same for nearly all sites.

  The intrusion scenarios considered in this analysis were developed based on the assumption of current human behavior to provide estimates of waste acceptance criteria in the form of concentration limits of radionuclides. Therefore, the issues associated with the recognized inability to predict the social behavior of populations far into the future were avoided.

- Particularly at sites located in humid regions, the estimated concentration limits for disposal of some radionuclides were determined by the analysis for the water pathway. At other sites and for many radionuclides, however, the water pathway was not important because the radionuclide travel time to the performance boundary either was much greater than the half-life of the radionuclide involved or was longer than the 10,000-year time of compliance assumed in the analysis. However, the estimates of permissible radionuclide concentrations in waste based on the peak concentrations, whenever they occur, have been calculated and are presented in the report.

  The modeling of the water pathway in this analysis is believed to be conservative for most sites. Therefore, in cases where a high concentration limit, or no limit, was estimated, a more sophisticated and rigorous analysis of the water pathway may not be warranted, provided performance measures similar to those assumed in this analysis were applied to future disposal facilities. On the other hand, in cases where a relatively low concentration limit for the water pathway was obtained (e.g., at the ORR and SRS sites), more refined and less conservative analyses, which take into account additional site-specific factors relevant to radionuclide transport in water, could be used to obtain more realistic calculated concentration limits for the water pathway based on additional site characterization data. Additionally, as site characterization continues and more information becomes available, additional exposure pathways might be identified, which could also result in changes to the concentration limits.

- The analysis for the water pathway clearly demonstrated that engineered barriers offer no significant long-term advantages for the disposal of wastes containing longer-lived

radionuclides. The primary advantage of engineered barriers is for the disposal of wastes containing shorter-lived radionuclides.

- The intrusion evaluation demonstrated that the permissible concentrations for medium-lived and longer-lived radionuclides were increased by up to three orders of magnitude at sites where the homesteader scenario was not credible (e.g., where waste was disposed of below grade at a depth sufficient to preclude this type of intrusion). However, a sufficiently thick vadose zone is required, a condition that generally occurs only at the arid sites.

- Through sensitivity analyses, the PE provided insights on key parameters (e.g., natural recharge and groundwater flow rates at a site, half-lives, and mobility of radionuclides) characterizing both the sites and the wastes and revealed the impacts of changes in these parameters on the estimated concentration limits for various radionuclides. The PE also showed that the degree of conservatism in the estimated concentration limits of radionuclides depend on the implicit assumptions in the transport models and scenarios as well as on values assigned to key input parameters.

- Indicator radionuclides were identified as effective surrogates for those radionuclides having similar properties and characteristics. The PE analysis showed that appropriately selected indicator radionuclides can be used in site-specific analyses of disposal facilities, thus reducing the analysis time and cost without resulting in significant additional uncertainties in the analyses.

- The PE methodology was demonstrated as a useful scoping-level tool which provides a readily available approach for identifying important transport and exposure pathways. The PE methodology can also be used to identify where more detailed site-specific water pathway transport analyses may be required to determine more realistic estimates of the concentration limits for specific radionuclides.

The PE methodology does not provide a substitute for the long-term performance assessments required by DOE Order 5820.2A for planned disposal facilities. It is likely that site-specific performance assessments for the water and atmospheric release pathways would differ from the results in the PE analysis; the magnitude of the difference depends primarily on the differences in the assumptions used in the analyses. Site-specific analyses of inadvertent intrusion also may differ from the results of the PE methodology in some cases.

The results of the PE or a site-specific performance assessment will not be the sole basis for decisions about waste disposal at particular DOE sites or within the DOE complex. A variety of additional factors need to be considered including, for example, the results of safety analyses for disposal facility operations, the degree to which a potential disposal site has already been contaminated by past operations or waste disposals, the benefits and costs associated with shipping waste from one site to another, and the issue of having many smaller disposal facilities at a variety of sites compared to having a smaller number of larger facilities at selected sites. Although adequate technical analyses are required for siting waste disposal facilities, economic and social concerns clearly will play an important role in their selection.

8. REFERENCES

Chu, M.S.Y., M.W. Kozak, J.E. Campbell, and B.M. Thompson, 1991. *A Self-Teaching Curriculum for the NRC/SNL Low-Level Waste Performance Assessment Methodology.* NUREG/CR-5539, SAND90-0585. Albuquerque, NM: Sandia National Laboratories.

Department of the Army, [1970] 1980. *Engineering and Design Laboratory Soils Testing.* EM-1110-2-1906, Reprint of Change 1. Washington, DC: U.S. Waterways Experiment Station for the Chief of Engineers.

DOE (Department of Energy), 1988a. "Low-Level Waste," Chap. III in *Radioactive Waste Management*, Order 5820.2A.

DOE (Department of Energy), 1988b. *Internal Dose Conversion Factors for Calculation of Dose to the Public.* DOE/EH-0071.

DOE (Department of Energy), 1988c. *External Dose-Rate Conversion Factors for Calculation of Dose to the Public.* DOE/EH-0070.

DOE (Department of Energy), 1992. *Integrated Data Base for 1991: U.S. Spent Fuel and Radioactive Waste Inventories, Projections, and Characteristics.* DOE/RW-0006, Rev. 7. Washington, DC: U.S. Department of Energy.

DOE (Department of Energy), 1994. *Framework for DOE Low-Level and Mixed Low-Level Waste Disposal: Current Overview.* DOE/ID-10484. Washington, DC: U.S. Department of Energy, Office of Waste Management.

Eckerman, K. F., and J. C. Ryman, 1993. *External Exposure to Radionuclides in Air, Water, and Soil.* Federal Guidance Report No. 12, EPA 402-R-93-081. Washington, DC: U.S. Environmental Protection Agency.

Eckerman, K. F., A. B. Wohlbarst, and A. C. B. Richardson, 1988. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion.* Federal Guidance Report No. 11, EPA-520/1-88-020. Washington, DC: U.S. Environmental Protection Agency.

EPA (Environmental Protection Agency), 1989. *Stabilization/Solidification of CERCLA and RCRA Wastes.* EPA/525/6-89/022. Cincinnati, OH: U.S. Environmental Protection Agency, Office of Research and Development.

EPA (Environmental Protection Agency), 1991. "40 CFR Parts 141, National Primary Drinking Water Regulations; Radionuclides, Proposed Rule," *Federal Register* 56, 33050.

EPA (Environmental Protection Agency), 1992. "Environmental Radiation Protection Standard for Management and Disposal of Spent Nuclear Fuel, High-Level and Transuranic Radioactive Waste," *Code of Federal Regulations 40, Part 191.* Washington, DC: Superintendent of Documents, U.S. Government Printing Office.

EPA (Environmental Protection Agency), 1993. "Part 61-National Emission Standards for Hazardous Air Pollutants," *Code of Federal Regulations 40, Part 61.* Washington, DC: Superintendent of Documents, U.S. Government Printing Office.

EPA (Environmental Protection Agency), 1994. "Federal Radiation Protection Guidance for Exposure of the General Public," proposed recommendations, *Federal Register* 59, 66414-66428 (December 23).

EPRI (Electric Power Research Institute), 1985. *A Review of Field-Scale Physical Solute Transport Processes in Saturated and Unsaturated Porous Media.* EPRI EA-4190. Norris, TN: Tennessee Valley Authority for the Electric Power Research Institute.

FFCAct, 1992. *Federal Facility Compliance Act (FFCAct) of 1992*, P.L. 102-386, October 1992.

Freeze, R. Allan, and John A. Cherry, 1979. *Groundwater.* Englewood Cliffs, NJ: Prentice-Hall, Inc.

Galya, D.P., 1987. *A Horizontal Plane Source Model for Ground-Water Transport.* Ground Water 25(6), 733-739.

GENE, 1989. *Nuclides and Isotopes: Chart of the Nuclides.* Eds. F. William Walker, Josef R. Parrington, and Frank Feiner. Fourteenth Edition. San Jose, CA: General Electric Company, Nuclear Energy Operations.

Glasser, F.P. and M. Adkins, 1994. *Cements in Radioactive Wastes.* MRS Bulletin, 19(12), 33-38.

Gruebel, Marilyn M., Robert D. Waters, Maryann B. Hospelhorn, and Margaret S.Y. Chu, eds., 1994. *Framework for DOE Low-Level and Mixed Low-Level Waste Disposal: Site Fact Sheets.* SAND94-2728. Albuquerque, NM: Sandia National Laboratories.

Guymon, G.L., 1994. *Unsaturated Zone Hydrology*, Englewood Cliffs, NJ: Prentice-Hall, Inc.

ICRP (International Commission on Radiological Protection), 1979. "Publication 30—Limits for Intakes of Radionuclides by Workers, Parts 1-3," *Annals of the International Commission on Radiological Protection*, Vols. 2-8.

ICRP (International Commission on Radiological Protection), 1986. "Publication 48—The Metabolism of Plutonium and Related Elements," *Annals of the International Commission on Radiological Protection,* Vol. 16, No. 2/3.

ICRP (International Commission on Radiological Protection), 1991. "Publication 60—1990 Recommendations of the International Commission on Radiological Protection," *Annals of the International Commission on Radiological Protection,* Vol. 21, No. 1-3.

Kincaid, C.T., J.W. Shade, G.A. Whyatt, M.G. Piepho, K. Rhoads, J.A. Voogd, J.H. Westsik, Jr., M.D. Freshley, K.A. Blanchard, and B.G. Lauzon, eds., 1993. *Performance Assessment of Grouted Double-Shell Tank Waste Disposal at Hanford.* WHC-SD-WM-EE-004, Rev. 0. U.S. Department of Energy, Pacific Northwest Laboratory and Westinghouse Hanford Company.

Magnuson, Swen O., Steven J. Majeras, Hoa D. Nguyen, Arthur S. Rood, Janice L. Sipos, Marilyn J. Case, Michael A. McKenzie-Carter, and Mary E. Donahue, 1992. *Radiological Performance Assessment for the Area 5 Radioactive Waste Management Site at the Nevada Test Site.* 11-1-2, Rev. 1. Idaho Falls, ID: U.S. Department of Energy, DOE Idaho Field Office.

Maheras, Steven J., Arthur S. Rood, Swen O. Magnuson, Mary E. Sussman, and Rajiv N. Bhatt, 1994. *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment.* EGG-WM-8773. Idaho Falls, ID: U.S. Department of Energy Office of Environmental Restoration and Waste Management, DOE Idaho Operations Office.

MMES (Martin Marietta Energy Systems, Inc.), EG&G Idaho, Inc., Westinghouse Hanford Company, and Westinghouse Savannah River Company, 1992. *Radiological Performance Assessment for the Z-Area Saltstone Disposal Facility (U).* WSRC-RP-92-1360, Rev. 0. Aiken, SC: Westinghouse Savannah River Company.

MMES (Martin Marietta Energy Systems, Inc.), EG&G Idaho, Inc., and Westinghouse Savannah River Company, 1994. *Radiological Performance Assessment for the E-Area Vaults Disposal Facility.* WSRC-RP-94-218, Rev. 0. Aiken, SC: Westinghouse Savannah River Company.

NAS (National Academy of Sciences), 1995. *Technical Bases for Yucca Mountain Standards.* Washington, DC: Committee on Technical Bases for Yucca Mountain Standards; Board on Radioactive Waste Management; Commission on Geosciences, Environment, and Resources; and National Research Council. National Academy Press.

NCRP (National Council on Radiation Protection and Measurements), 1993. *Limitation of Exposure to Ionizing Radiation.* NCRP Report No. 116. Bethesda, MD: National Council on Radiation Protection and Measurements.

NRC (Nuclear Regulatory Commission), 1982. "Licensing Requirements for Land Disposal of Radioactive Waste", *Final Environmental Impact Statement on 10 CFR Part 61.* NUREG-0945.

Office of the President, 1987. "Radiation Protection Guidance to Federal Agencies for Occupational Exposure; Approval of Environmental Protection Agency Recommendations," *Federal Register* 52, 2822.

ORNL (Oak Ridge National Laboratory), 1994. *Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6.* ORNL-6783. Oak Ridge, TN: U.S. Department of Energy, Oak Ridge National Laboratory.

Resource Conservation and Recovery Act (RCRA) of 1976, P.L. 94-580, Stat. 2795, 1976.

Rosinger, E. L. J., and K. K. R. Tremaine, 1980. *GARD2: A Computer Program for Geosphere Systems Analysis.* AECL-6432. Atomic Energy of Canada, Ltd.

Winter, G. and A.H. Nilson, 1979. *Design of Concrete Structures, Ninth Edition.* New York: McGraw-Hill Book Company.

Wood, D. E., R. U. Curl, D. R. Armstrong, J. R. Cook, M. R. Dolenc, D. C. Kocher, K. W. Owens, E. P. Regnier, G. W. Roles, R. R. Seitz, and M. I. Wood, 1994. *Performance Assessment Task Team Progress Report.* DOE/LLW-157, Rev. 1. Idaho Falls, ID: Idaho National Engineering Laboratory.

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.* WHC-EP-0645. Richland, WA: Westinghouse Hanford Company for the U.S. Department of Energy, Office of Environmental Restoration and Waste Management.

APPENDIX A:

MEMBERS OF THE PE PROCESS TEAM

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APPENDIX B:

CONCENTRATION REDUCTION FACTORS

FOR THE WATER PATHWAY

NOMENCLATURE

a	Length of the areal source (m)
A_1	Facility plan area
A_2	Area at the water table
C_{gw}	Ambiant groundwater concentration
C_T	Total concentration based on the total mass flux
CRF_{Water}	Ratio of the leachate to total groundwater concentration
d	Distance from the edge of the areal source (m)
d_m	Mixing depth in groundwater (m)
i	Natural recharge of water through local soils at disposal site (cm/y)
I_a	Rate of water flowing through the facility at time a
I_b	Rate of water flowing through the facility at time b
K_d	Radionuclide-specific distribution coefficient for the geologic media (mL/g)
l	Distance between the facility and the water table
L	Vadose zone thickness (m)
$\dot{m}$	Leachate mass flux (g/y)
n	Aquifer porosity (dimensionless)
q_f	Rate of water flowing through the disposal facility (m/y)
q_w	Ambient moisture content in the vadose zone (mL/cm^3)
Q_L	Leachate flow
C_L	Leachate concentration
Q_{gw}	Groundwater flow
Q_T	Total flow
R	Retardation factor (dimensionless)
t_a	Time of leachate collection system failure
t_b	Time of failure of engineered barriers
t_{crit}	Critical half-life
t_{cva}	Arbitrary travel time through entire vadose zone
t_{wv}	Water travel time in the vadose zone (y)
θ_W	Ambient moisture content

APPENDIX B:
CONCENTRATION REDUCTION FACTOR FOR THE WATER PATHWAY

The PE made several assumptions in the generic conceptual model to estimate the concentration reduction factor for the water pathway (see Section 5.3.1 of this volume). These assumptions pertain to

- dilution due to mixing of leachate and groundwater flows,
- one-dimensional transport in the vadose zone (no spreading),
- longitudinal dispersion for a continuous source,
- transverse dispersion for a continuous source,
- water travel time in the vadose zone,
- water travel time in the saturated zone, and
- performance of engineered barriers, recharge, and vadose zone travel time.

B.1 DILUTION DUE TO MIXING OF LEACHATE AND GROUNDWATER FLOWS

Equation 5-11 in Section 5.3.1.2 of this volume is the result of a mass flux mixing procedure outlined below. The procedure assumes that the leachate flow, Q_L, has a leachate concentration, C_L, and that the groundwater flow, Q_{gw}, has an ambient groundwater concentration, C_{gw}. From mass balance, the total flow is the sum of the two flows

$$Q_T = Q_L + Q_{gw} \tag{B-1}$$

and the total concentration based on the total mass flux is

$$C_T = \left(Q_L C_L + Q_{gw} C_{gw}\right) / Q_T \quad . \tag{B-2}$$

Assuming that the ambient background concentration is zero simplifies the resulting total concentration to

$$C_T = Q_L C_L / Q_T = \dot{m} / Q_T \tag{B-3}$$

where $\dot{m}$ is the leachate mass flux (g/y). Because CRF_{Water} is defined as the ratio of the leachate to total groundwater concentration, in terms of flows, CRF_{Water} is defined as

$$CRF_{Water} = Q_L + Q_{gw} / Q_L \tag{B-4}$$

which is the form used in Equation 5-11 in Section 5.3.1.2.

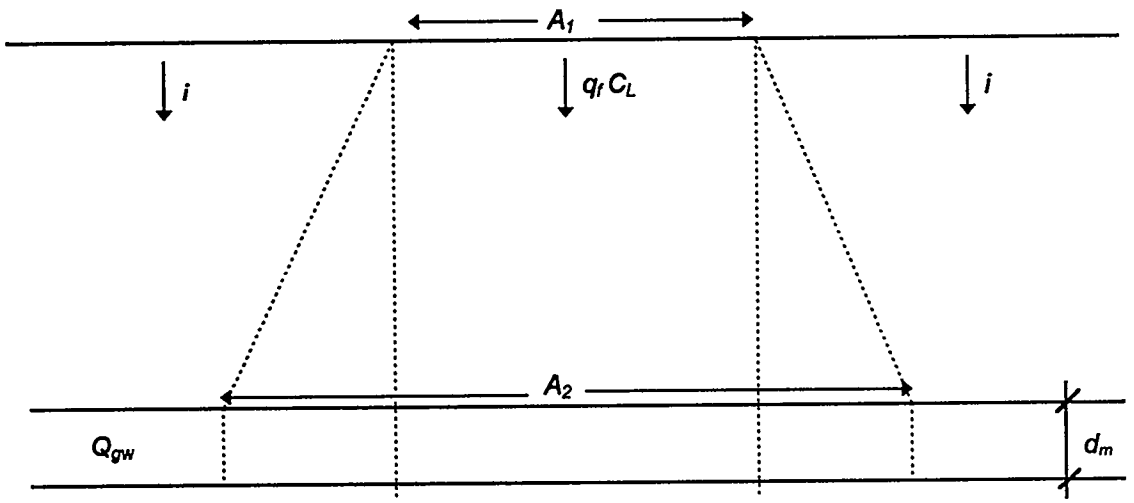


Figure B-1. Effect of one-dimensional transport in the vadose zone.

B.2 ONE-DIMENSIONAL TRANSPORT IN THE VADOSE ZONE (NO SPREADING)

The effect of the one-dimensional transport assumption in the vadose zone is illustrated in two dimensions in Figure B-1. In the assumed case of one-dimensional transport, the mass flux reaching the groundwater, applied uniformly over the facility plan area, A_1, is

$$\dot{m} = A_1 q_f C_L \quad . \tag{B-5}$$

At long times when q_f equals i, the contaminant q_f is allowed to spread horizontally as it moves downward, so that the area at the water table is A_2; the mass flux reaching groundwater is

$$\dot{m} = A_2 \, q_f \, C_L^{*} \, . \tag{B-6}$$

Equating Equations B-5 and B-6, C_L^{*} is found to be proportionally less than C_L by the ratio of A_1 and A_2:

$$C_L^{*} = C_L \frac{A_1}{A_2} \quad . \tag{B-7}$$

The assumption of contaminant spreading in the vadose zone has no effect on the groundwater concentration downstream of the mixing zone. As seen in Equation B-3, the downstream groundwater concentration due to mixing of leachate and groundwater equals the mass flux, $\dot{m}$, of leachate divided by the combined flow of groundwater and leachate. The mass flux, $\dot{m}$, in Equations B-3, B-5, and B-6 is identical due to conservation of mass. By assuming no spreading in the vadose zone, the calculations are simplified and require fewer input parameters with no change in results.

During the period when engineered barriers are assumed to be effective, i can be greater than q_f. The no-spreading assumption is conservative in this case.

B.3 LONGITUDINAL DISPERSION FOR A CONTINUOUS SOURCE

The downstream concentration profile at breakthrough for the step function for a continuous contaminant source at time t for one-dimensional flow and transport is illustrated in Figure B-2(a).

In the absence of transverse dispersion, the shape of the concentration profile is determined solely by the assumed longitudinal dispersion. Peak downstream concentration for a non-decaying solute is equal to the original concentration at the source. When the source release is continuous for a finite time of sufficient duration, the peak downstream concentration, as shown in Figure B-2(b), is still equal to the original concentration at the source.

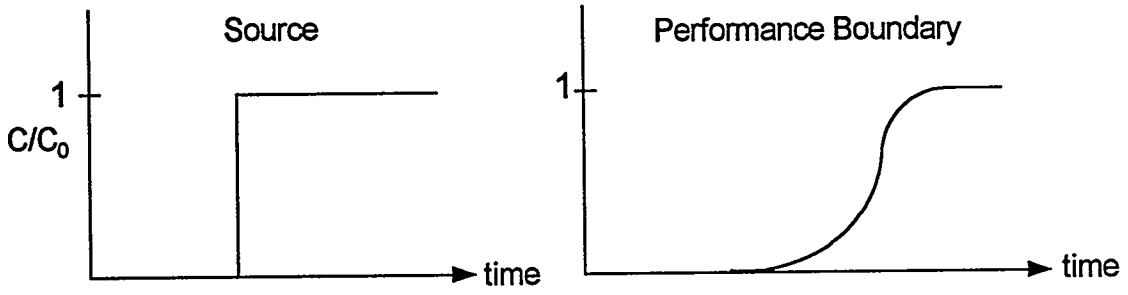
Only for source releases of limited duration will the peak downstream concentration be attenuated by dispersion, as shown in Figure B-3.

The duration of time for a finite source to be well represented by a continuous source is a function of the pore velocity and distribution coefficient. The continuous source assumption provides results that are either correct or conservative relative to a release of finite duration.

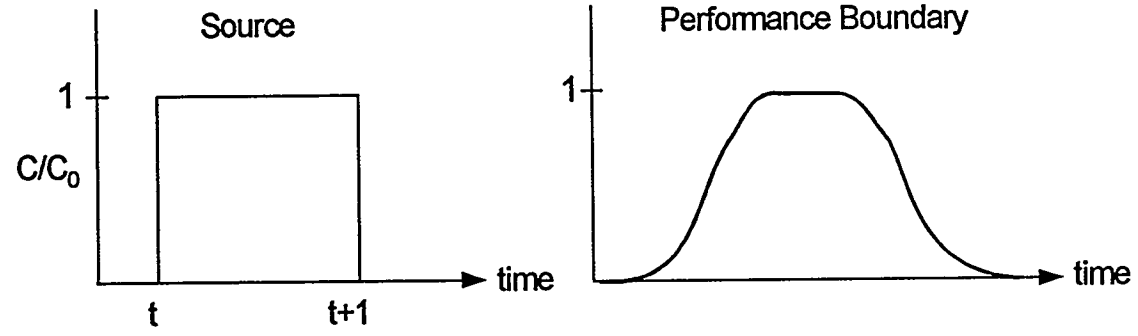
B.4 TRANSVERSE DISPERSION FOR A CONTINUOUS SOURCE

The effects of lateral transverse dispersion on a continuous source in two dimensions along the axis of the direction of flow are shown in Figure B-4.

The edges of the square contaminant wave are attenuated by transverse dispersion, and the centerline peak concentration is also attenuated. The amount of attenuation is a function transverse dispersivity and distance. Because the distance from the edge of the disposal facility to the performance boundary is 100 m, peak centerline attenuation by transverse dispersion is expected to be minor. Therefore, neglecting the effects of lateral transverse dispersion is conservative relative to incorporating it.



(a) At breakthrough



(b) At peak downstream concentration

Figure B-2. Downstream concentration profiles for a continuous contaminant source (a) at breakthrough and (b) at peak downstream concentration.

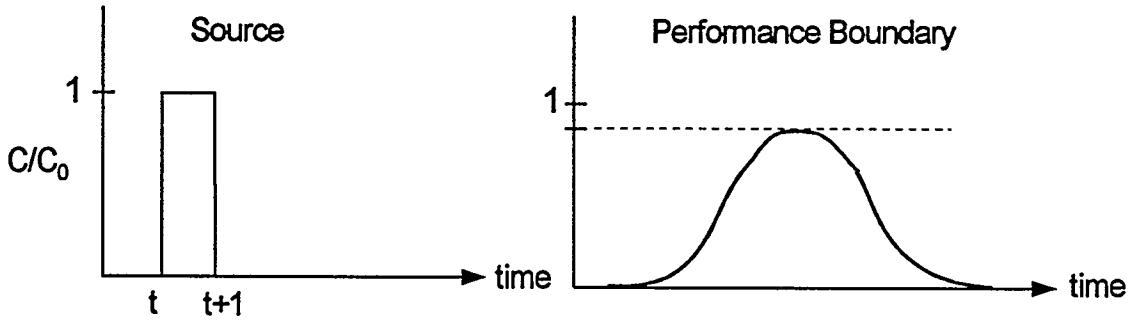


Figure B-3. Downstream concentration profile for a source release of limited duration.

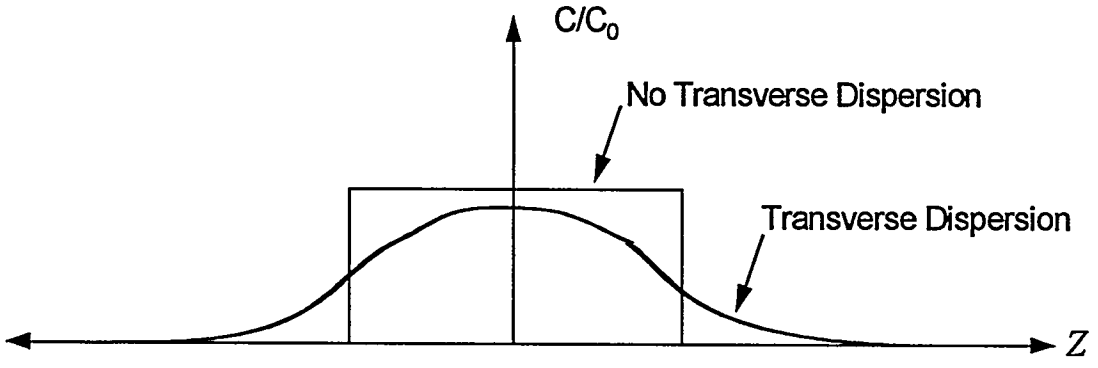


Figure B-4. Effects of lateral transverse dispersion on a continuous source.

B.5 WATER TRAVEL TIME IN THE VADOSE ZONE

Water travel time in the vadose zone is used to estimate the time of contaminant arrival at the performance boundary and is calculated as

$$t_{wv} = \frac{l\theta_w}{q_f} \tag{B-8}$$

where

l is the distance between the facility and the water table;
θ_w is the ambient moisture content; and
q_f is the rate of water flowing through the facility.

The low permeability cover and liners associated with a disposal facility will reduce recharge and may lower the ambient moisture content. The reduction in moisture content can be estimated by the empirical van Genuchten relationship. The effect of lowering the ambient moisture content is a decrease in travel time through the vadose zone. Ignoring this effect is not conservative but is only important for highly mobile (zero K_d), short half-life radionuclides at sites with thick vadose zones.

For radionuclides with K_ds larger than zero, the retardation effect overwhelms the moisture content effects. Decay of long-lived radionuclides is fairly insensitive to small changes in travel time. Thin vadose zones translate to fast travel times, and small changes to large travel times are insignificant.

Additionally, thick vadose zones are generally associated with arid sites where natural recharge is very small and similar in magnitude to water movement through a disposal facility. At these sites (e.g., Nevada Test Site), the permissible waste concentrations will likely be limited by intrusion or atmospheric releases. Therefore, using the ambient moisture content in the PE results in only minor lack of conservatism.

B.6 WATER TRAVEL TIME IN THE SATURATED ZONE

The concept that the time required to reach the maximum concentration at a fixed distance downstream from an areal source is a function of the water travel time from the center of the source of dimensions a by a is illustrated in Figure B-5.

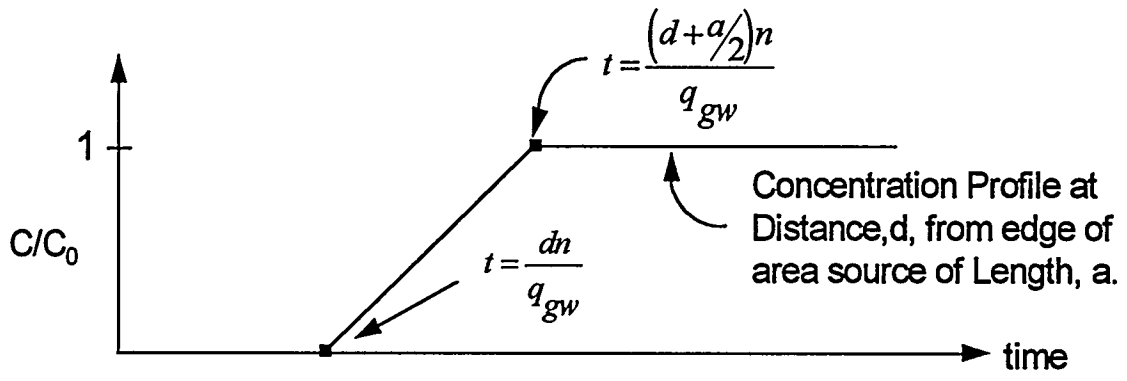


Figure B-5. Time of maximum concentration at a fixed distance as a function of water travel time from the center of the source.

As seen in Figure B-5, the maximum concentration is reached at time

$$t_{ws} = \frac{(d + a/2)n}{q_{gw}} \tag{B-9}$$

where

d is the distance from the edge of the areal source (m),
a is the length of the areal source (m),
q_{gw} is the groundwater Darcy velocity (m/y), and
n is the aquifer porosity (dimensionless).

Therefore, to estimate the time of arrival of the peak concentration at a distance, d, from the edge of a disposal facility, the distance must be measured to the center of the facility.

B.7 PERFORMANCE OF ENGINEERED BARRIERS, RECHARGE, AND VADOSE ZONE TRAVEL TIME

An assumption used in the PE is that no radionuclides leach from the disposal facility while the leachate collection system is intact. As shown in Figure B-6, when the leachate collection system fails at time t_a, radionuclides begin to leach from the facility at rate I_a. At time t_b, the engineered barriers are assumed to fail and movement of water through the facility occurs at rate I_b. As described in Chapter 5 of this volume, the rate of water movement (I_a) at t_a will be less than or equal to the rate of water movement (I_b) at t_b.

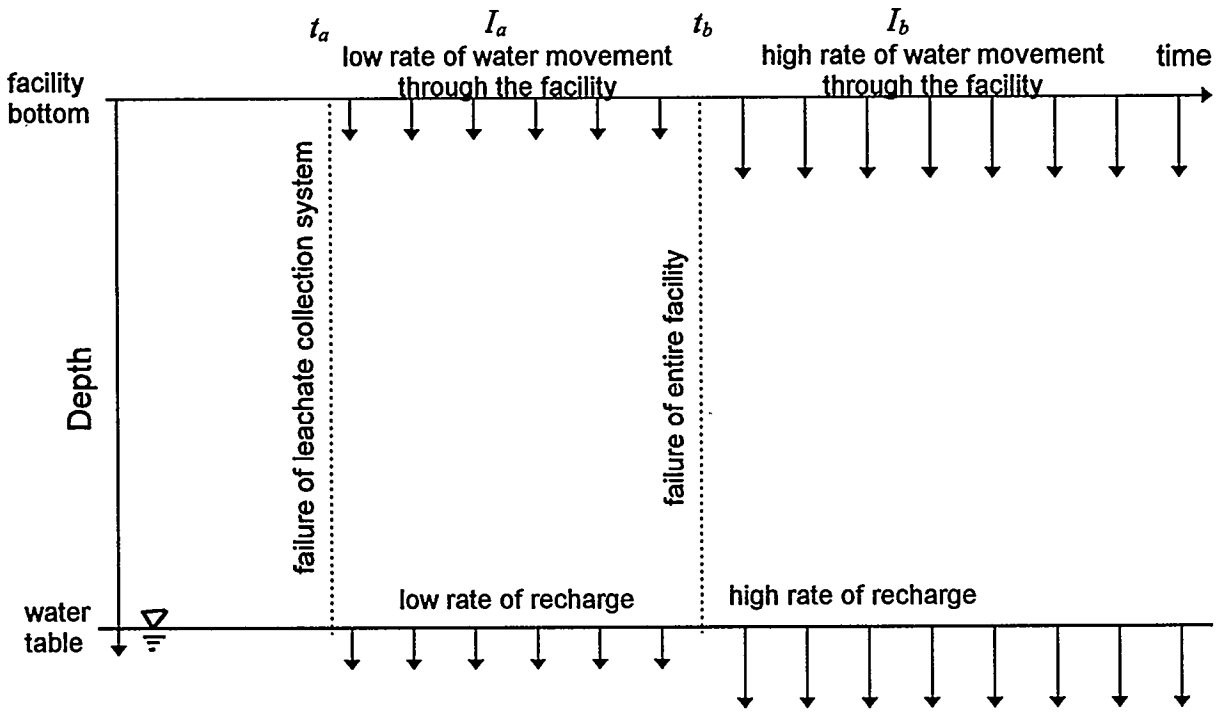


Figure B-6. Leach rates of radionuclides from the disposal facility.

For each radionuclide, an arbitrary travel time through the entire vadose zone, t_{cva}, is calculated at the lower rate of water movement (see Figure B-7) as

$$t_{cva} = \frac{R\,L\,\theta_W}{I_a} \tag{B-10}$$

where

R is the retardation factor (-),
L is the vadose zone thickness (m), and
θ_W is the ambient moisture content in the vadose zone (mL/cm^3).

If the contaminant arrives at the water table before t_b, then a check is performed to determine which recharge is used to determine the CRF_{Water} based upon the relationship between travel time and radioactive decay. Thus, the "critical half-life", t_{crit}, is defined as

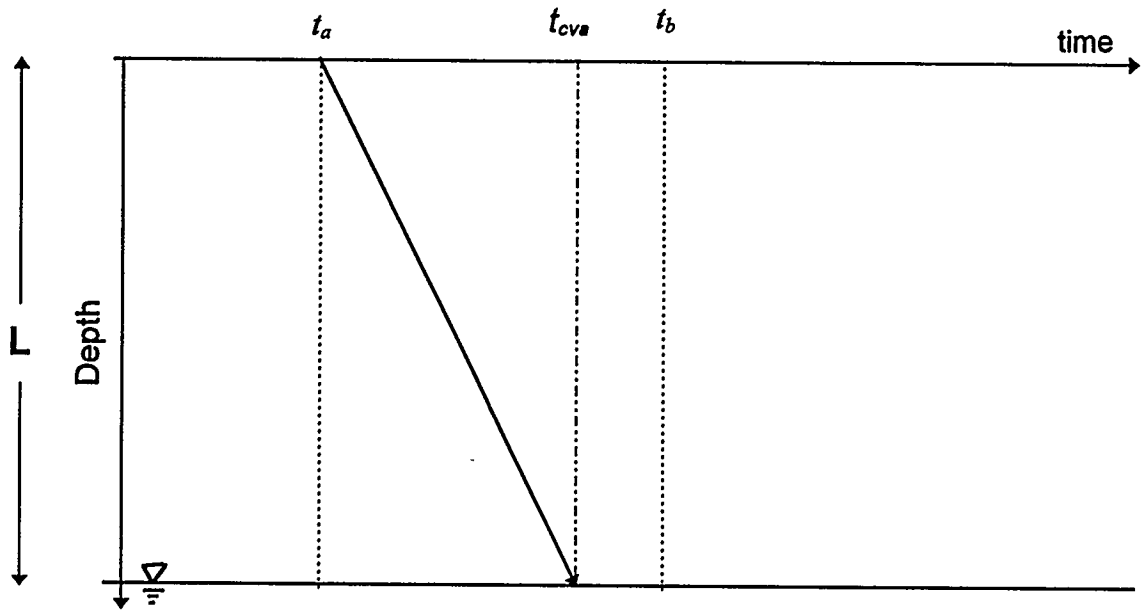


Figure B-7. Calculation of travel time through the vadose zone.

$$t_{crit} = [t_b - (t_a + t_{cva})]\frac{\ln 2}{\ln (I_b/I_a)} \qquad \text{(B-11)}$$

If the half-life of the radionuclide is shorter than the critical half-life, then the early rate, I_a, controls and the travel time through the vadose zone, t_{cv}, is equal to t_{cva}. For this case, I_a is used to determine the CRF_{Water}. If the half-life of the radionuclide is longer than the critical half-life, then the late rate, I_b, controls and t_{cv} is equal to t_b - t_a. For this case, I_b is used to determine the CRF_{Water}.

The leachate may not arrive at the water table by t_b at sites with thick vadose zones and low natural recharge (see Figure B-8). In this situation, the leachate released at t_a travels a distance L_1 at rate I_a and a distance L_2 at rate I_b. The travel time through the vadose zone is the sum of the time required to travel L_1 and L_2, and the CRF_{Water} is determined by the higher rate.

Specifically:

$$t_{cv} = t(L_1) + t(L_2) \qquad \text{(B-12)}$$

The distances are defined as

$$L = L_1 + L_2 \qquad \text{(B-13)}$$

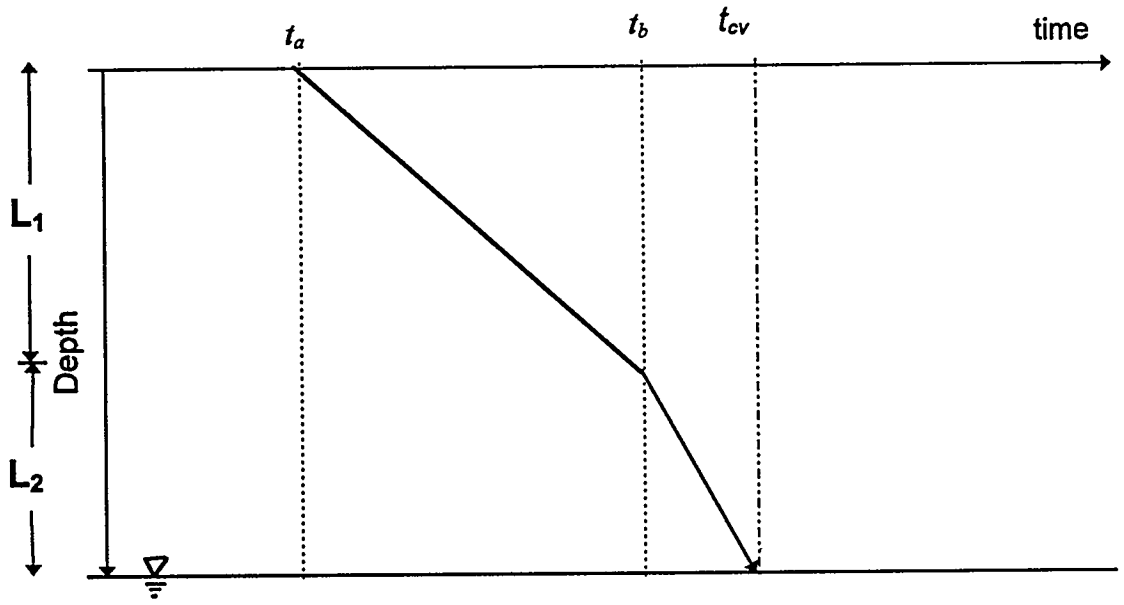


Figure B-8. Calculation of travel time at sites with thick vadose zones and low natural recharge.

$$t(L_1) = (t_b - t_a) = \frac{L_1 R \theta_W}{I_a} \tag{B-14}$$

$$t(L_2) = \frac{L_2 R \theta_W}{I_b} \tag{B-15}$$

Substituting equations B-14 and B-15 into B-13 results in

$$t_{cv} = (t_b - t_a) + \frac{L_2 R \theta_W}{I_b} \tag{B-16}$$

Then, substituting for L_2

$$t_{cv} = (t_b - t_a) + (L - L_1)\frac{R \theta_W}{I_b} \tag{B-17}$$

Finally, substituting for L_1

$$t_{cv} = (t_b - t_a) + [L - (t_b - t_a)\frac{I_a}{R \theta_W}]\frac{R \theta_W}{I_b} \tag{B-18}$$

APPENDIX C:

ATMOSPHERIC PATHWAY ANALYSIS

NOMENCLATURE

A_D	Atmospheric dispersion term specifying the concentration in ambient air at the performance boundary produce by unit flux density leaving the soil at the disposal facility ($\mu Ci/m^3$ per ($\mu Ci/m^2$-s)
AH	Absolute humidity of the atmosphere (kg/m^3),
$C_{a,H-3}$	Concentration of H-3 in air ($\mu Ci/m^3$)
$C_{a,i}$	Concentration of radionuclide *i* in air ($\mu Ci/m^3$)
$C_{air,C}$	Concentration of natural carbon in air (kg/m^3)
$C_{air,C-14}$	Concentration of C-14 in air ($\mu Ci/m^3$)
C_{Atm}	Radionuclide concentration in the ambient air resulting from the air concentration at the facility boundary ($\mu Ci/m^3$)
$C_{(b,m),i}$	Concentration of radionuclide *i* in beef (b) ($\mu Ci/kg$) or milk (m) ($\mu Ci/L$)
C_{FB}	Radionuclide flux density out of the surface soil
C_{pw}	Volatile radionuclide concentration in the pore water ($\mu Ci/m^3$)
C_{SS}	Radionuclide cencentration in soil above waste disposal facility resulting from concentration in waste ($\mu Ci/m^3$)
$C_{b,i}$	Concentration of radionuclide *i* in beef from cattle that consumed contaminated vegetation (in $\mu Ci/kg$)
$C_{m,i}$	Concentration of radionuclide *i* in milk from dairy cows that consumed contaminated vegetation (in $\mu Ci/L$)
C_v	Volatile radionuclide vapor concentration ($\mu Ci/m^3$)
$C_{v,C-14}$	Concentration of C-14 in vegetation ($\mu Ci/kg$)
$C_{v,H-3}$	Concentration of H-3 in vegetation ($\mu Ci/kg$)
$C_{v,i}$	Concentration of radionuclide *i* in vegetation ($\mu Ci/kg$)
C_{Waste}	Radionuclide concentration in the waste ($\mu Ci/m^3$)
CRF_{Diff}	Concentration reduction factor for environmental transport by soil diffusion in atmospheric pathway (dimensionless)
CRF_{Disp}	Concentration reduction factor for environmental transport by dispersion in air in the atmospheric pathway (dimensionless)
d	Depth of the surface soil
D	Diffusion coefficient in air
$DCF_{Ing,i}$	Internal dose conversion factor for ingestion of radionuclide *i* ($rem/\mu Ci$)
$DCF_{Inh,i}$	Internal dose conversion factor from inhalation for radionuclide *i* ($rem/\mu Ci$)
EPA	Environmental Protection Agency
$F_{b,i}$	Ratio of equilibrium concentration of radionuclide *i* in meat to daily intake by beef cattle ($\mu Ci/kg$ in meat per $\mu Ci/d$ intake)
f_{Cv}	Fraction of natural carbon in vegetation (dimensionless)
$F_{m,i}$	Ratio of equilibrium concentration of radionuclide *i* in milk to daily intake by dairy cows ($\mu Ci/L$ in milk per $\mu Ci/d$ intake)
f_{wv}	Fraction of vegetation that is water (dimensionless).
$H_{(b,m),i}$	Annual dose from radionuclide *i* in beef (b) or milk (m) (rem/y)
$H_{Inh,i}$	Annual dose from inhalation of radionuclide *i* in air (rem/y)
$H_{T,i}$	Annual dose from atmospheric releases of radionuclide *i*
$H_{v,i}$	Annual dose from radionuclide *i* in vegetation (rem/y)
INEL	Idaho National Engineering Laboratory
IR_a	Intake rate of air (adult inhalation rate) (m^3/y)
$IR_{(b,m)}$	Adult intake rate of beef (b) (kg/y) or milk (m) (kg/L), and
IR_v	Adult intake rate of vegetables (kg/y)
$IR_{v(cow)}$	Consumption rate of vegetation by beef cattle or dairy cows (kg/y)
ISCLT2	Industrial Source Complex Long Term [Dispersion Model Version] 2
J	Radionuclide flux density through the soil above the waste disposal facility ($\mu Ci/m^2$-s)
k	Henry's Law constant for CO_2
n	Soil porosity
N_{CO2}	Number of moles in 1 g of CO_2
N_{H2O}	Number of moles in 1 cm^3 of liquid water
NRC	Nuclear Regulatory Commission
NTS	Nevada Test Site
ORR	Oak Ridge Reservation
p	Partial pressure of CO_2 in air (atm)
r	Ratio of water vapor present in dry air to that in the liquid phase
$R_{wv/wa}$	Concentration ratio of H-3 in vegetation water to H-3 in atmospheric water (dimensionless)
s	Percent saturation in the soil void space
SRS	Savannah River Site
t	Diffusion transport time for volatile radionuclides
x	Cover thickness above the waste disposal facility (m)
χ	Unit mole fraction

APPENDIX C:
ATMOSPHERIC PATHWAY ANALYSIS

The discussion below outlines the proposed method by which the atmospheric transport portion of the 16 performance evaluation analyses is conducted. The calculation of the two atmospheric concentration reduction factors is detailed in Sections C.1 and C.2, the contaminant transport time is discussed in Section C.3, and the proposed modeling method that is used to determine the atmospheric dispersion term is discussed in Section C.4. Section C.5 presents the exposure analysis for the atmospheric pathway and develops pathway dose conversion factors.

The PE analysis considers only the volatile radionuclides H-3 and C-14. The following method is derived from performance assessment documents written for low-level radioactive waste disposal facilities at Hanford (Kincaid et al., 1993), INEL (Maheras et al., 1994), NTS (Baer et al., 1994), ORR (ORNL, 1994), and SRS (MMES et al., 1994). The method borrows heavily from these documents and has been generalized for use at all sixteen sites. As much site-specific data are used in the calculations as possible.

The method determines two atmospheric concentration reduction factors:

- CRF_{Diff} is defined as the concentration reduction that occurs from the top of the disposal facility to the soil surface as the radionuclide diffuses upward.
- CRF_{Disp} is defined as the concentration reduction that occurs as the radionuclide is emitted to the atmosphere and dispersed downwind from the waste disposal facility boundary to the performance boundary.

C.1 CRF_{Diff} CALCULATION

CRF_{Diff} is calculated as:

$$CRF_{Diff} = \frac{C_{Waste}}{C_{SS}} \qquad \text{(C-1)}$$

where

C_{Waste} is the radionuclide concentration in the waste ($\mu Ci/m^3$), and
C_{SS} is the radionuclide concentration in the soil surface above the waste disposal unit resulting from the concentration in the waste ($\mu Ci/m^3$).

Volatile radionuclides are assumed to be transported to the soil surface by diffusion in the vapor phase (MMES et al., 1994). The vapor flux is assumed to be a first-order, linear process, and flux is assumed to occur from the top of the waste disposal unit to the soil surface. Therefore, the volatile radionuclide vapor concentration, C_v ($\mu Ci/m^3$), at the top of the waste disposal unit is approximated by:

$$C_v = J\left(\frac{x}{D}\right) \qquad \text{(C-2)}$$

where

J is the radionuclide flux density through the soil above the waste disposal facility ($\mu Ci/m^2$-s),

D is the H-3 diffusion coefficient in air reported as 2.39×10^{-5} m^2/s or the $^{14}CO_2$ diffusion coefficient in air reported as 1.40×10^{-5} m^2/s (CRC, 1981), and

x is the cover thickness above the waste disposal facility (m).

For H-3, the relation between its concentration in the water vapor and in the liquid water bound in the waste must be determined before C_W can be calculated. This can be accomplished by determining the ratio of water density in air to that in the liquid phase. If the air is saturated with water at 10°C, then the density of water vapor in dry air, or absolute humidity, is 9.2 grams of water vapor per cubic meter of dry air (MMES et al., 1994). Assuming the density of liquid water is 1×10^6 grams per cubic meter, the ratio of water vapor present in dry air to that in the liquid phase, r, is:

$$r = \frac{9.2 \times 10^{-6}\ g/m^3\ (\text{water vapor})}{1\ g/m^3\ (\text{liquid water})} \qquad \text{(C-3)}$$

The relation between the concentration of C-14 in the air and in the liquid water bound in the waste can be calculated using Henry's Law. This law is a linear, first-order relationship derived by approximating equilibrium conditions between dissolved gas with a particular concentration in liquid water and the same gas with a particular concentration in the air adjacent to the liquid water. All of the C-14 in the disposal facility is assumed to be $^{14}CO_2$. For use in Henry's Law, the unit mole fraction, χ, of $^{14}CO_2$ in the liquid water is determined as:

$$\chi = \frac{N_{CO2}\ per\ g\ of\ CO_2}{N_{H2O}\ per\ g\ of\ H_2O} \qquad \text{(C-4)}$$

where

N_{CO2} is the number of moles in 1 gram of CO_2, and

N_{H2O} is the number of moles in 1 cubic meter of liquid water.

Assuming the density of liquid water as 1.0×10^6 grams per cubic meter, χ is equal to 4.1×10^{-7}. The partial pressure of CO_2 in the air, p (atm), is calculated using Henry's Law:

$$p = k\chi \qquad \text{(C-5)}$$

where

k is the Henry's Law constant for CO_2. At 10°C, k is equal to 1040 atm/mole fraction (Foust et al., 1960 as cited in Cooper and Alley, 1986).

Substituting the values for the unit mole fraction and Henry's Law constant for CO_2 at 10°C, p is calculated to be 4.3×10^{-4} atm. If the air is at atmospheric pressure, the concentration of CO_2 in the air is approximately equal to 4.3×10^{-4} g/m^3, and therefore, the ratio, r, of CO_2 in the air to that dissolved in the water is:

$$r = \frac{4.3 \times 10^{-4}\ \text{g/m}^3\ (\text{CO}_2\ \text{in air})}{1\ \text{g/m}^3\ (\text{CO}_2\ \text{dissolved in water})} \quad \text{(C-6)}$$

Because H-3 is assumed to be completely bound in the pore water and C-14 is assumed to be dissolved as $^{14}CO_2$, the volatile radionuclide concentration in the pore water, C_{pw} (μCi/m^3) is:

$$C_{pw} = \frac{C_v}{r} \quad \text{(C-7)}$$

The radionuclide concentration in the waste, C_{Waste}, is related to the C_{pw} as:

$$C_{Waste} = C_{pw}ns \quad \text{(C-8)}$$

where

n is the soil porosity, and
s is the percent saturation in the soil void space.

Substituting Equation C-7 into Equation C-8 yields:

$$C_{Waste} = \frac{C_v ns}{r} \quad \text{(C-9)}$$

and substituting Equation C-2 into Equation C-9 gives:

$$C_{Waste} = \frac{Jxns}{Dr} \quad \text{(C-10)}$$

The flux density out of the soil surface is assumed to be equal to the flux density through the soil, J. Therefore, C_{SS} is calculated as:

$$C_{SS} = \frac{Jd}{D} \quad \text{(C-11)}$$

where

d is the depth of the surface soil. The value for d is assumed to be 0.01 m for all sites (Maheras et al., 1994).

Substituting Equation C-10 and C-11 into Equation C-1 gives:

$$CRF_{Diff} = \frac{xns}{rd} \quad \text{(C-12)}$$

C.2 CRF_{Disp} CALCULATION

CRF_{Disp} is calculated as:

$$CRF_{Disp} = \frac{C_{SS}}{C_{Atm}} \quad \text{(C-13)}$$

where

C_{SS} is the radionuclide concentration in the soil surface above the waste disposal unit ($\mu Ci/m^3$), and

C_{Atm} is the radionuclide concentration in the ambient air resulting from the air concentration at the facility boundary ($\mu Ci/m^3$).

C_{SS} is calculated using Equation C-11.

The radionuclide flux density out of the surface soil is assumed to be emitted directly into the atmosphere where it is mixed with the ambient air flowing above the facility. Therefore, C_{FB} is determined by:

$$C_{FB} = JA_D \quad \text{(C-14)}$$

where

A_D is the atmospheric dispersion term specifying the concentration in ambient air at the performance boundary produced by unit flux density leaving the soil at the disposal facility ($\mu Ci/m^3$ per $\mu Ci/m^2$-s).

A_D is the maximum annual average value among a set of receptors located at the performance boundary; it can be obtained using a Gaussian air dispersion model (see Section C.5).

Equation C-11 and Equation C-14 are substituted into Equation C-13, yielding:

$$CRF_{Disp} = \frac{d}{A_D D} \quad \text{(C-15)}$$

C.3 TRANSPORT TIME TO THE RECEPTORS

Assuming that the diffusive velocity, v_d, is uniform and that it is approximated as one-dimensional, first-order, and linear, the diffusion transport time, t, for volatile radionuclides, is given by (Fisher et al., 1979):

$$t = \frac{x^2}{D} \quad \text{(C-16)}$$

where

D is the diffusion coefficient in air (m^2/s).

Once airborne, the transport time to receptors located at the performance boundary downwind would probably take less than 100 seconds. Therefore, there would not be sufficient time for appreciable radionuclide decay during both the mixing phase and the dispersion phase of transport.

C.4 MIXING AND ATMOSPHERIC DISPERSION TERM DETERMINATION

The atmospheric dispersion term, A_D ($\mu Ci/m^3$ per $\mu Ci/m^2$-s), is estimated using the Industrial Source Complex-Version 2 Long Term Air Dispersion Model (ISCLT2) (EPA, 1992a). A_D is defined as the maximum annual average value calculated for a set of receptors located 100 m from the waste disposal facility. ISCLT2 is the EPA's refined air dispersion model for calculating long-term (annual average) atmospheric concentrations in simple terrain. GENII and AIRDOS-PC are two Gaussian dispersion models that are specifically approved for use in modeling affects of radionuclide emissions; however, both models have features that go well beyond the need of the PE analysis and would be considerably more cumbersome to use than ISCLT2. Because these three models incorporate the same basic dispersion equations, differences in the concentration estimates generated by the models would not be significant (less than an order of magnitude).

The area of the waste disposal facility is an important input into the model. For the general tumulus and shallow trench designs, flux areas are the same for all sixteen sites. For those sites that have mixed waste disposal plans, site-specific waste disposal facility areas are used. In order to improve the accuracy of the model with receptors at close distances, the surface area of the disposal facility is divided into 256 equal squares. Note the length of the smaller squares is 1/16th the length of the side of the facility area. Site-specific meteorological data are also used. Because the area flux is assumed to occur at ground-level, terrain is modeled as flat. This is an upper-bound assumption that results in the maximum ground-level receptor concentrations.

C.5 EXPOSURE ANALYSIS AND SCENARIO DOSE CONVERSION FACTORS

The performance objective for atmospheric releases (10 mrem/y) includes doses from all potential exposure pathways associated with such releases. The annual doses for the two volatile radionuclides of interest (H-3 and C-14) are calculated from four exposure pathways:

- inhalation of airborne radionuclides,
- ingestion of vegetation exposed to airborne radionuclides (i.e., airborne-contaminated vegetation),
- ingestion of beef from cattle consuming airborne-contaminated vegetation, and
- ingestion of milk from cows consuming airborne-contaminated vegetation.

No external doses are expected since the beta particles emitted by these two radionuclides have very low energies; in fact, these two radionuclides have external dose conversion factors equal to zero (DOE, 1988). The total dose from the atmospheric transport pathway is the sum of the doses from each of the four exposure pathways listed above. The equations used to calculate each exposure pathway dose are described below and the parameter values used in these equations are listed in Table C-1. This methodology is based on a conservative (i.e., higher) specific activity model presented in NRC Regulatory Guide 1.109 (NRC, 1977). Note that

because this is a pathway analysis, radioactive decay is not included in the calculations. To determine dose or, conversely, permissible waste limits, radioactive decay at the time of exposure must be taken into account.

Table C-1. Parameter Values Used to Determine Doses from All Potential Exposure Pathways Associated with Atmospheric Emissions from Waste Disposal Facilities

		Nuclide-Specific Value		
Parameter	**Value**	**H-3**	**C-14**	**Source**
IR_a (adult average) (m^3/y)	8000			ICRP 1975
$DCF_{inh,i}$ (rem/μCi)		6.4E-05	2.20E-05[b]	EPA 1988
AH (kg/m^3)	0.0092			MMES et al. 1994
$R_{wv/wa}$ (dimensionless)	0.5			NRC 1977
f_{wv} (dimensionless)	0.75			NRC 1977
$C_{air,C}$ (kg/m^3)	0.00016			NRC 1977; Napier et al. 1988
f_{Cv} (dimensionless)	0.11			NRC 1977
IR_v (kg/y)	90[a]			MMES et al. 1994
$DCF_{ing,i}$ (rem/μCi)		6.4E-05	2.1E-03	EPA 1988
$F_{b,i}$ (d/kg)		1.2E-02	3.1E-02	NRC 1977
$IR_{v(cow)}$ (kg/y)	5694			EPA 1992b
$F_{m,i}$(d/L)		1.0E-02	1.2e-02	NRC 1977
IR_b (kg/y)	85			EPA 1992b
IR_m (L/y)	112			EPA 1992b

a Assumes approximately half of an individual's vegetable intake is from locally-grown, contaminated vegetation.
b Carbon as CO_2.

The dose from direct inhalation of volatiles is calculated using the following equation:

$$H_{inh,i} = C_{a,i} * IR_a * DCF_{inh,i} \quad \text{(C-17)}$$

where:

$H_{inh,i}$ is the annual dose from inhalation of radionuclide i in air (rem/y),
$C_{a,i}$ is the concentration of radionuclide i in air (μCi/m^3),
IR_a is the intake rate of air (adult inhalation rate) (m^3/y), and (rem/μCi).

To calculate the doses due to H-3 contamination in food, it is assumed that the source of the contamination is the air surrounding the vegetation. The concentration in the vegetation is

based on the amount of water in the vegetation and the amount of H-3 that would be in the plant water. The equation for calculating the concentration of H-3 in vegetation is as follows:

$$C_{v,H\text{-}3} = (C_{a,H\text{-}3} / AH) * R_{wv/wa} * f_{wv} \quad \text{(C-18)}$$

where:

$C_{v,H\text{-}3}$ is the concentration of H-3 in vegetation (μCi/kg),

$C_{a,H\text{-}3}$ is the concentration of H-3 in air (μCi/m^3),

AH is the absolute humidity of the atmosphere (kg/m^3),

$R_{wv/wa}$ is the concentration ratio of H-3 in vegetation water to H-3 in atmospheric water (dimensionless), and

f_{wv} is the fraction of vegetation that is water (dimensionless).

The concentration of H-3 in air, $C_{a,H\text{-}3}$, divided by the absolute humidity, AH, is equivalent to the concentration in water. This, when multiplied by the fraction of water in the vegetation, f_{wv} and the concentration ratio of H-3 in vegetation water to atmospheric water, $R_{wv/wa}$, equals the concentration of H-3 in the vegetation.

To estimate the doses from the ingestion of C-14 contaminated fool, it is assumed that the source of the contamination is the air surrounding the vegetation. It is also assumed that the ratio of C-14 to the natural carbon in vegetation is the same as the ratio of C-14 to natural carbon in the atmosphere surrounding the vegetation. For airborne releases, it is also assumed that plants obtain all their carbon from airborne CO_2 and that animals obtain all their carbon through ingestion of plants (NRC, 1977, p. 1.109-26; Napier, et al., 1988, p. 4.86).

The equation for calculating the concentration of C-14 in vegetation from contaminated air is:

$$C_{v,C\text{-}14} = (C_{air,C\text{-}14} / C_{air,C}) * f_{Cv} \quad \text{(C-19)}$$

where:

$C_{v,C\text{-}14}$ is the concentration of C-14 in vegetation (μCi/kg),

$C_{air,C\text{-}14}$ is the concentration of C-14 in air (μCi/m^3),

$C_{air,C}$ is the concentration of natural carbon in air (kg/m^3), and

f_{Cv} is the fraction of natural carbon in vegetation (dimensionless).

The concentration in vegetation is then used to calculate the dose from ingestion of contaminated vegetation. As shown in Table C-1, approximately 50% of the exposed person's vegetable intake is assumed to involve contaminated vegetation (MMES et al., 1994). The resulting dose can be estimated as follows:

$$H_{v,i} = C_{v,i} * IR_v * DCF_{ing,i} \quad \text{(C-20)}$$

where:

$H_{v,i}$ is the annual dose from radionuclide i in vegetation (rem/y),

$C_{v,i}$ is the concentration of radionuclide i in vegetation (μCi/kg),

IR_v is the adult intake rate of vegetables (kg/y), and
$DCF_{ing,i}$ is the internal dose conversion factor for ingestion of radionuclide i (rem/μCi).

The concentration in vegetation is also used to determine the concentration in beef and milk that will be consumed by humans. It is assumed that 100% of the cow's vegetation consumption is from grazing on fresh pasture grass contaminated with airborne radionuclides and that the animal grazes 365 days of the year. The concentrations in beef and in cow's milk are estimated as follows:

$$C_{b,i} = C_{v,i} * F_{b,i} * IR_{v(cow)} / (365\ d/y) \quad \text{(C-21)}$$

$$C_{m,i} = C_{v,i} * F_{m,i} * IR_{v(cow)} / (365\ d/y) \quad \text{(C-22)}$$

where:

$C_{b,i}$ is the concentration of radionuclide i in beef from cattle that consumed contaminated vegetation (in μCi/kg),
$C_{m,i}$ is the concentration of radionuclide i in milk from dairy cows that consumed contaminated vegetation (in μCi/L),
$C_{v,i}$ is the concentration of radionuclide i in vegetation consumed by beef cattle or dairy cows,
$F_{b,i}$ is the ratio of equilibrium concentration of radionuclide i in meat to daily intake by beef cattle (μCi/kg in meat per μCi/d intake),
$F_{m,i}$ is the ratio of equilibrium concentration of radionuclide i in milk to daily intake by dairy cows (μCi/L in milk per μCi/d intake), and
$IR_{v(cow)}$ is the consumption rate of vegetation by beef cattle or dairy cows (kg/y).

The annual radiation doses from the ingestion of beef and milk are calculated by multiplying the concentration in each medium by the human intake rate and the radionuclide-specific internal dose conversion factor:

$$H_{b,i} = C_{b,i} * IR_b * DCF_{ing,i} \quad \text{(C-23)}$$

$$H_{m,i} = C_{m,i} * IR_m * DCF_{ing,i} \quad \text{(C-24)}$$

where:

$H_{(b,m),i}$ is the annual dose from radionuclide i in beef (b) or milk (m) (rem/y),
$C_{(b,m),i}$ is the concentration of radionuclide i in beef (b) (μCi/kg) or milk (m) (μCi/L),
$IR_{(b,m)}$ is the adult intake rate of beef (b) (kg/y) or milk (m) (kg/L), and
$DCF_{ing,i}$ is the internal dose conversion factor for ingestion of radionuclide i (rem/μCi).

The annual dose from atmospheric releases of radionuclide i, $H_{T,i}$, is the sum of the annual doses from the four exposure pathways described above (i.e., inhalation of contaminated air and ingestion of contaminated vegetables, beef, and milk):

$$H_{T,i} = H_{inh,i} + H_{v,i} + H_{b,i} + H_{m,i} \qquad \text{(C-25)}$$

For an assumed air concentration of 1 $\mu Ci/m^3$, the above equations can be used to determine the annual dose for each of the two radionuclides. The values in Table C-1, when incorporated into the above equations, yield scenario dose conversion factors of 8.33×10^2 mrem/y per $\mu Ci/m^3$ for H-3 and 2.20×10^5 mrem/y per $\mu Ci/m^3$ for C-14. Table C-2 illustrates the contribution of each pathway to the overall dose.

Table C-2. All Pathways Dose Resulting from Exposure to Air Contaminated at a Concentration of 1 $\mu Ci/m^3$ of H-3 or C-14

Pathway	Radionuclide			
	H-3		C-14	
	mrem/y	Percent of Total Dose	mrem/y	Percent of Total Dose
Direct inhalation	512	61	190	<1
Consumption of contaminated vegetables	234	28	129,900	59
Consumption of contaminated beef	42	5	59,380	27
Consumption of contaminated milk	45	6	30,390	14
TOTAL	833 mrem/y[a]	100 %	219,860 mrem/y[b]	100 %

[a]Scenario dose conversion factor is 8.33×10^2 mrem/y per $\mu Ci/m^3$.
[b]Scenario dose conversion factor is 2.20×10^5 mrem/y per $\mu Ci/m^3$.

References for Appendix C

Baer, T. A. et al., 1994. *Second Performance Assessment Iteration of the Greater Confinement Disposal Facility at the Nevada Test Site.* SAND93-0089. Albuquerque, NM: Sandia National Laboratories.

Cooper, C. D., and F. C. Alley, 1986. *Air Pollution Control: A Design Approach.* Prospect Heights, IL: Waveland Press, Inc.

CRC, 1981. *CRC Handbook of Chemistry and Physics.* Eds. Robert C. Weast and Melvin J. Astle. Boca Raton, FL: CRC Press, Inc.

DOE (U. S. Department of Energy), 1988. *Internal Dose Conversion Factors for Calculation of Dose to the Public*, DOE/EH-0071, July.

EPA (U. S. Environmental Protection Agency), 1988. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion*, Federal Guidance Report No. 11, Office of Radiation Programs, EPA 520/1-88-020, September.

EPA (U.S. Environmental Protection Agency), 1992a. *User's Guide for the Industrial Source Complex (ISC2) Dispersion Models, Volume I: User Instructions.* EPA Publication No. EPA-450/4-92-008a. Research Triangle, NC: U.S. Environmental Protection Agency.

EPA (U. S. Environmental Protection Agency), 1992b. *User's Guide for CAP88-PC Version 1.0*, Air and Radiation, 402-B-92-001, Las Vegas, Nevada.

Fisher, H. B., E. J. List, R. C. Koh, J. Imberger, and N. H. Brooks, 1979. *Mixing in Inland and Coastal Waters.* New York, NY: Academic Press.

Foust et al., 1960. *Principles of Unit Operations.* New York, NY: John Wiley & Sons, Inc.

ICRP (International Commission on Radiological Protection), 1975. *International Commission on Radiological Protection, Task Group Report on Reference Man,* ICRP Publication No. 23, Pergamon Press, New York.

Kincaid, C. T. et al., 1993. *Performance Assessment of Grouted Double-Shell Tank Waste Disposal at Hanford, Vol. 1.* WHC-SD-WM-EE-004, Rev. 0. Richland, WA: Pacific Northwest Laboratory and Westinghouse Hanford Company.

Maheras, S. J. et al., 1994. *Radioactive Waste Management Complex Low-Level Waste Radiological Performance Assessment.* EGG-WM-8773. Idaho Falls, ID: Idaho National Engineering Laboratory.

MMES (Martin Marietta Energy Systems, Inc.), EG&G Idaho, Inc., and Westinghouse Savannah River Company, 1994. *Radiological Performance Assessment for the E-Area Vaults Facility (U).* WSRC-RP-94-218. Aiken, SC: Westinghouse Savannah River Company.

Napier et al., 1988. *GENII - The Hanford Environmental Radiation Dosimetry Software System, Volume 1: Conceptual Representation*, Battelle, Pacific Northwest Laboratory, Hanford, Washington, December.

NRC (U. S. Nuclear Regulatory Commission), 1977. *Regulatory Guide 1.109. Calculation of Annual doses to Man from Routine Releases of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I.*

ORNL (Oak Ridge National Laboratory), 1994. *Performance Assessment for Continuing and Future Operations at Solid Waste Storage Area 6.* ORNL-6783. Oak Ridge, TN: Oak Ridge National Laboratory.

APPENDIX D:

INTRUDER SCENARIO EXPOSURE PATHWAYS

NOMENCLATURE

B_v	Plant-to-soil concentration ratio for each radionuclide [(μCi/kg fresh weight in vegetation)/(μCi/kg dry weight in soil)]
C_w	Concentration of radionuclide in waste disposal unit (μCi/m^3),
DCF_{ext}	Dose-rate conversion factor for external exposure [(rem/y)/(μCi/m^3)]
DCF_{ing}	Dose conversion factor for ingestion (rem/μCi)
DCF_{inh}	Dose conversion factor for inhalation (rem/μCi)
f_{Decay}	Fraction of radionuclide remaining in waste after decay (dimensionless)
f_{et}	Fraction of year exposed
f_g	Geometric correction factor (fraction of waste versus non-waste material in disposal unit)
f_m	Mixing correction factor (i.e., mixing of waste with uncontaminated soil/material upon excavation/drilling)
f_s	Shielding factor of home during indoor exposure
H_{eg}	Annual external dose from working in a garden with contaminated soil (rem/y)
H_{eh}	Annual external dose from living in a house (rem/y)
H_{ig}	Annual inhalation dose from suspended soils while working in a garden (rem/y)
H_{ih}	Annual inhalation dose from suspended soils while in the home (rem/y)
$H_{I(drill)}$	Post-drilling intruder dose (rem/y)
$H_{I(home)}$	Homesteader intruder dose (rem/y)
$H_{I(x)}$	Total intruder annual dose for scenario x (rem/y),
H_s	Annual soil ingestion dose associated with vegetable intake (rem/y)
H_v	Annual vegetable ingestion dose from contaminated soil uptake in the plant (rem/y)
IR_a	Inhalation rate of air (m^3/y)
IR_s	Soil consumption (intake) rate (kg/y)
IR_v	Vegetable consumption (intake) rate (kg fresh weight/y)
L_a	Atmospheric mass loading of surface soil (kg/m^3)
PE	Performance evaluation
P_s	Bulk density of soil (kg/m^3)
$SCDF$	Scenario dose conversion factor [(rem/y)/(μCi/m^3)]
$SDCF_{(i)}$	Scenario dose conversion factor [(rem/y)/(μCi/m^3)] for exposure pathway i

APPENDIX D:
INTRUDER SCENARIO EXPOSURE PATHWAYS

The generic intruder equation for a specific radionuclide is as follows:

$$C_w = H_{I(x)} / (\Sigma SDCF_{(i)} * f_{Decay}) \quad \text{(D-1)}$$

where

C_w is the concentration of radionuclide in waste disposal unit ($\mu Ci/m^3$),
$H_{I(x)}$ is the total intruder annual dose for scenario x (rem/y),
$SDCF_{(i)}$ is the scenario dose conversion factor [(rem/y)/($\mu Ci/m^3$)] for exposure pathway i, adjusted for factors applicable to specific exposure pathway or scenario (e.g., fraction of radionuclide in waste accounting intake rate, fraction of waste mixed with uncontaminated soil, fraction of time exposed, plant-to-soil concentration ratios, etc.), and
f_{Decay} is the fraction of radionuclide remaining in waste after decay, e^{-lt}, ($l = \ln 2/t_{1/2}$), dimensionless.

The total intruder doses [$H_{I(x)}$] are made up of doses from several exposure pathways. The homesteader intruder dose [$H_{I(home)}$] and the post-drilling intruder dose [$H_{I(drill)}$] are the sums of doses from the following exposure pathways:

$$H_{I(home)} = H_v + H_s + H_{eh} + H_{eg} + H_{ig} + H_{ih} \quad \text{(D-2)}$$

$$H_{I(drill)} = H_v + H_s + H_{eg} + H_{ig} \quad \text{(D-3)}$$

where

$H_{I(x)}$ is the total intruder annual dose for scenario x,
H_v is the annual vegetable ingestion dose from contaminated soil uptake in the plant,
H_s is the annual soil ingestion dose associated with vegetable intake,
H_{eh} is the annual external dose from living in house,
H_{eg} is the annual external dose from working in garden with contaminated soil,
H_{ig} is the annual inhalation dose from suspended soils while working in garden, and
H_{ih} is the annual inhalation dose from suspended soils while in the home.

Note that water pathways have been eliminated as explained in Section 5.3.3.

Although the equation for an exposure pathway dose is the same for different scenarios, they may not be equal because some factors used to calculate these terms would be different for each scenario (see Table D-1).

Table D-1. Input Parameters for Intruder Scenarios

FACTOR		VALUE	UNITS	FACTOR VARIABILITY	SOURCE
DCF_{ing} DCF_{inh} DCF_{ex}	dose conversion factors for ingestion, inhalation, and external exposure	See Table D-2	rem/μCi (ingestion and inhalation) and (rem/y)/(μCi/m^3) (external)	nuclide specific, exposure route specific	EPA, 1988; EPA, 1993
IR_v	intake rate for vegetables grown in own garden	90	kg/y	no variability (constant) - (possibly site specific)	WSRC, 1994
IR_s	intake rate for soil	0.037	kg/y	no variability (constant) - (possibly site specific)	WSRC, 1994
IR_a	inhalation rate of average adult	8000	m^3/y	no variability (constant)	ICRP, 1975; WSRC, 1994
P_s	soil density	1400	kg/m^3	no variability (constant)	WSRC, 1994
B_v	soil-to-plant concentration ratio	See Table D-2	unitless (μCi/kg fresh wt vegetation per μCi/kg dry wt. soil)	chemical specific	Baes et al., 1984; WSRC, 1994
L_a	atmospheric loading of surface soil	$1x10^{-7}$ (garden) $1x10^{-8}$ (home)	kg/m^3	exposure pathway specific	WSRC, 1994
f_g	geometric correction factor (fraction of waste in disposal unit)	0.5 (tumulus) 1.0 (trench)	unitless	technology and scenario specific	WSRC, 1994
f_m	soil mixing correction factor	0.2 (homesteader) 0.02 (post-drilling	unitless	scenario specific	WSRC, 1994
f_{et}	fraction of year exposed	0.5 (home) 0.01 (garden)	unitless	exposure pathway specific	WSRC, 1994
f_s	shielding factor	0.7	unitless	exposure pathway specific	NRC 1977; WSRC, 1994
f_{Decay}	decay factor (time-dependent)	based on decay coefficient (and thus half-life) and time to intrusion	unitless	nuclide specific, scenario specific	

Table D-2. Radionuclide-Specific Constants for Intruder Scenario Calculations (Part 1 of 3)

NUCLIDE	DOSE CONVERSION FACTORS					SOIL-TO-PLANT CONCENTRATION RATIOS
	Ingestion	Inhalation		External		Bv
	rem/µCi	Class	rem/µCi	(rem/y)/(µCi/m3)		µCi/kg fresh wt. vegetation per µCi/kg dry wt. soil
				15 cm soil depth	infinite depth	
H-3	6.40E-05		6.40E-05			
C-14 (organic)	2.09E-03		2.09E-03			
C-14 (as CO2)			2.35E-05			
Al-26	1.46E-02	D	7.96E-02	9.0E-03	1.1E-02	2.80E-04
Si-32	2.18E-03	Y	1.01E+00			3.01E-02
Cl-36	3.03E-03	W	2.19E-02			3.01E+01
K-40	1.86E-02	D	1.24E-02			2.37E-01
Co-60	2.69E-02	Y	2.19E-01	8.5E-03	1.0E-02	3.01E-03
Ni-59	2.10E-04	D	1.32E-03			2.58E-02
Ni-63	5.77E-04	D	3.10E-03			2.58E-02
Se-79	8.70E-03	W	9.84E-03			1.08E-02
Sr-90	1.42E-01	D	2.39E-01			1.08E-01
*Y-90	1.08E-02	Y	8.44E-03			2.58E-03
Zr-93	1.66E-03	D	3.21E-01			2.15E-04
Nb-93m	5.22E-04	Y	2.92E-02			2.15E-03
Nb-94	7.14E-03	Y	4.14E-01			2.15E-03
Tc-99	1.46E-03	D	1.02E-03			6.45E-01
Pd-107	1.49E-04	Y	1.28E-02			1.72E-02
Ag-108m	7.62E-03	Y	2.83E-01			4.30E-02
*Ag-108						
Cd-113m	1.61E-01	D	1.53E+00			6.45E-02
Sn-121m	1.55E-03	D	6.51E-03	1.2E-06	1.2E-06	2.58E-03
Sn-126	1.95E-02	D	8.73E-02	9.2E-05	9.2E-05	2.58E-03
*Sb-126	1.02E-02	D	4.70E-03	9.5E-03	1.1E-02	1.29E-02
*Sb-126m				5.2E-03	5.8E-03	
I-129	2.76E-01	D	1.74E-01	8.1E-06	8.1E-06	2.15E-02
Cs-135	7.07E-03	D	4.55E-03			1.29E-02
Cs-137	5.00E-02	D	3.19E-02			1.29E-02
*Ba-137m	3.40E-03			2.0E-03	2.3E-03	6.45E-03
Ba-133	3.40E-03	D	7.81E-03			6.45E-03
Sm-151	3.89E-04	W	3.00E-02			1.72E-03
Eu-152	6.48E-03	W	2.21E-01	3.8E-03	4.4E-03	1.72E-03
Eu-154	9.55E-03	W	2.86E-01	4.1E-03	4.8E-03	1.72E-03
Pb-210	5.37E+00	D	1.36E+01			3.87E-03
*Po-210	1.90E+00	W	8.58E+00			1.72E-04

Table D-2. Radionuclide-Specific Constants for Intruder Scenario Calculations (Part 2 of 3)

NUCLIDE	DOSE CONVERSION FACTORS					SOIL-TO-PLANT CONCENTRATION RATIOS
	Ingestion	Inhalation		External		Bv
	rem/µCi	Class	rem/µCi	(rem/y)/(µCi/m3)		µCi/kg fresh wt. vegetation per µCi/kg dry wt. soil
				15 cm soil depth	infinite depth	
Ra-226	1.32E+00	W	8.58E+00			6.45E-04
*Pb-214	6.25E-04	D	7.81E-03	7.8E-04	8.4E-04	3.87E-03
*Bi-214	2.83E-04	D	6.59E-03	5.1E-03	6.1E-03	2.15E-03
Ra-228	1.44E+00	W	4.77E+00			6.45E-04
Th-229	3.53E+00	Y	1.73E+03	2.0E-04	2.0E-04	3.66E-05
*Ra-225	3.85E-01			6.9E-06	6.9E-06	6.45E-04
*Ac-225	1.11E-01			3.9E-05	4.0E-05	1.51E-04
*Fr-221				9.2E-05	9.6E-05	
*Bi-213				4.4E-04	4.8E-04	
*Tl-209				6.8E-03	8.1E-03	
Th-230	5.48E-01	Y	2.62E+02			3.66E-05
*Ac-228				3.2E-03	3.7E-03	
Th-232	2.73E+00	Y	1.15E+03			3.66E-05
*Th-228	3.96E-01	Y	3.42E+02			3.66E-05
*Ra-228	1.44E+00	W	4.77E+00			6.45E-04
*Ra-224	3.66E-01					6.45E-04
*Pb-212	4.55E-02			4.2E-04	4.4E-04	3.87E-03
*Ac-228				3.2E-03	3.7E-03	
*Bi-212				6.3E-04	7.3E-04	
*Tl-208				1.1E-02	1.4E-02	
Pa-231	1.06E+01	W	1.28E+03	1.1E-04	1.2E-04	1.08E-04
*Ac-227	1.41E+01	D	6.70E+03			1.51E-04
*Ra-223	6.59E-01			3.6E-04	3.8E-04	6.45E-04
*Th-227				3.1E-04	3.3E-04	
*Pb-211				1.7E-04	1.9E-04	
*Bi-211				1.5E-04	1.6E-04	
*Tl-207				1.1E-05	1.2E-05	
U-232	1.31E+00	Y	6.59E+02			1.72E-03
*Th-228	3.96E-01	Y	3.42E+02			3.66E-05
*Ra-224	3.66E-01					6.45E-04
*Pb-212	4.55E-02					3.87E-03
U-233	2.89E-01	Y	1.35E+02			1.72E-03
U-234	2.83E-01	Y	1.32E+02			1.72E-03
U-235	2.66E-01	Y	1.23E+02	4.4E-04	4.5E-04	1.72E-03
*Th-231				2.3E-05	2.3E-05	
U-236	2.69E-01	Y	1.25E+02			1.72E-03

Table D-2. Radionuclide-Specific Constants for Intruder Scenario Calculations (Part 3 of 3)

NUCLIDE	DOSE CONVERSION FACTORS					SOIL-TO-PLANT CONCENTRATION RATIOS
	Ingestion	Inhalation		External		Bv
	rem/μCi	Class	rem/μCi	(rem/y)/(μCi/m3)		μCi/kg fresh wt. vegetation per μCi/kg dry wt. soil
				15 cm soil depth	infinite depth	
U-238	2.55E-01	Y	1.18E+02			1.72E-03
*Th-234	1.37E-02			1.5E-05	1.5E-05	3.66E-05
*Pa-234m				4.9E-05	5.6E-05	
*Pa-234				6.3E-03	7.2E-03	
Np-237	4.44E+00	W	5.40E+02	4.9E-05	4.9E-05	4.30E-03
*Pa-233				6.0E-04	6.4E-04	
Pu-238	3.20E+00	W	3.92E+02			1.94E-05
Pu-239	3.54E+00	W	4.29E+02			1.94E-05
Pu-240	3.54E+00	W	4.29E+02			1.94E-05
Pu-241	6.85E-02	W	8.25E+00			1.94E-05
Pu-242	3.36E+00	W	4.11E+02			1.94E-05
Pu-244	3.32E+00	W	4.03E+02			1.94E-05
*Np-240m				1.1E-03	1.3E-03	
Am-241	3.64E+00	W	4.44E+02	2.7E-05	2.7E-05	1.08E-04
Am-243	3.62E+00	W	4.40E+02	8.9E-05	8.9E-05	1.08E-04
*Np-239				4.6E-04	4.7E-04	
Cm-243	2.51E+00	W	3.07E+02	3.5E-04	3.6E-04	6.45E-06
Cm-244	2.02E+00	W	2.48E+02			6.45E-06
Cm-245	3.74E+00	W	4.55E+02	2.1E-04	2.1E-04	6.45E-06
Cm-246	3.70E+00	W	4.51E+02			6.45E-06
Cm-247	3.42E+00	W	4.14E+02	1.0E-03	1.1E-03	6.45E-06
*Pu-243				4.9E-05	5.0E-05	
Cm-248	1.36E+01	W	1.65E+03			6.45E-06
Cf-249	4.74E+00	W	5.77E+02	1.1E-03	1.2E-03	6.45E-06
Cf-250	2.13E+00	W	2.62E+02			6.45E-06
Cf-251	4.85E+00	W	5.88E+02	3.2E-04	3.3E-04	6.45E-06

NOTE: Where data are not provided in the table, it is assumed that the corresponding exposure pathway is negligible. For example, external exposures for many radionuclides (e.g., plutonium isotopes) are negligible due to the lack of emissions of high-energy photons.

(*) denotes radiologically significant short-lived decay products that are assumed to be in secular equilibrium with the parent radionuclide (WSRC, 1994).

Internal (ingestion and inhalation) dose conversion factors (DCFs) are based on EPA (1988) and external DCFs are based on WSRC (1994) and EPA (1993).

Soil-to-plant concentrations ratios (Bv) are based on values from Baes et al. (1984) (also used in WSRC, 1994). They are based on μCi/kg fresh weight in vegetation per μCi/kg dry weight in soil. These are based on ratios reported on the basis of dry weight of vegetation (Fig. 2.2 in Baes et al., 1984) multiplied by a factor of 0.43 to convert fresh weight of vegetation (Baes et al., 1984).

External exposures while living in the home are assumed to be from radionuclides uniformly distributed in infinite soil thickness (WSRC, 1994, Table A.4-5, p. A-58).

"Class" refers to the lung clearance class. The class that gives the highest dose was chosen because little is known of expected chemical forms and solubilities in waste or soil. The only exceptions are Sr-90 and Tc-99 (Class D), which are relatively soluble, and Th-xx (Class Y), which is expected to be highly insoluble (WSRC, 1994, p. A-50).

If one were to do a forward calculation to determine each exposure pathway dose, the following equations would be employed:

H_v is the annual vegetable ingestion dose

$$H_v = C_w * DCF_{ing} * f_{Decay} * IR_v * B_v * f_g * f_m / P_s \quad \text{(D-4)}$$

H_s is the annual soil ingestion dose

$$H_s = C_w * DCF_{ing} * f_{Decay} * IR_s * f_g * f_m / P_s \quad \text{(D-5)}$$

H_{eh} is the annual external dose from living in house

$$H_{eh} = C_w * DCF_{ext} * f_{Decay} * f_g * f_s * f_{et} \quad \text{(D-6)}$$

H_{eg} is the annual external dose from working in garden with contaminated soil

$$H_{eg} = C_w * DCF_{ext} * f_{Decay} * f_g * f_m * f_{et} \quad \text{(D-7)}$$

H_{ig} is the annual inhalation dose from suspended soils while working in garden

$$H_{ig} = C_w * DCF_{inh} * f_{Decay} * IR_a * f_g * f_m * f_{et} * L_a / P_s \quad \text{(D-8)}$$

H_{ih} is the annual inhalation dose from suspended soils while in the home

$$H_{ih} = C_w * DCF_{inh} * f_{Decay} * IR_a * f_g * f_{et} * L_a / P_s \quad \text{(D-9)}$$

where:

C_w is the concentration of radionuclide in exhumed waste (mCi/m^3),
DCF_{ing} is the dose conversion factor for ingestion (rem/μCi),
DCF_{ext} is the dose-rate conversion factor for external exposure [(rem/y)/(μCi/m^3)],
DCF_{inh} is the dose conversion factor for inhalation (rem/μCi),
f_{Decay} is the decay factor - fraction of initial inventory of radionuclide in disposal unit (based on time elapsed before initial exposure, i.e., time of intrusion); (e^{-lt}; $l = \ln 2/t_{1/2}$),
IR_v is the vegetable consumption (intake) rate (kg fresh weight/y),
IR_s is the soil consumption (intake) rate (kg/y),
IR_a is the inhalation rate of air (m^3/y),
f_g is the geometric correction factor (fraction of waste versus non-waste material in disposal unit),
f_m is the mixing correction factor (i.e., mixing of waste with uncontaminated soil/material upon excavation/drilling),
B_v is the plant-to-soil concentration ratio for each radionuclide [(μCi/kg fresh weight in vegetation)/(μCi/kg dry weight in soil)],
P_s is the bulk density of soil (kg/m^3),

f_s is the shielding factor of home during indoor exposure,
f_{et} is the fraction of year exposed, and
L_a is the atmospheric mass loading of surface soil (kg/m^3).

Values for these factors are given in Tables D-1 and D-2.

For each exposure pathway dose, the scenario dose conversion factor (*SDCF*) is the product of all the factors on the right sides of the equation except for the concentration in the waste (C_w) and the decay factor (f_{decay}). For example, the *SDCF* for the vegetable ingestion pathway would be:

$$SDCF = DCF_{ing} * IR_v * B_v * f_m / P_s \qquad \text{(D-10)}$$

The total *SDCF* for an intruder scenario is the sum of each *SDCF* for each exposure pathway applicable to an intruder scenario, as depicted in Equation D-1.

Some of the factors used to calculate the *SDCF*s are constant but may only be constant for specific exposure pathways, disposal technologies, sites, etc. Table D-1 lists the factors required to calculate the intruder doses identified above and lists what variability they will have when conducting the analyses for the PEs. That is, some of these factors will be constant for all sites and all scenarios, while others may vary depending on disposal technology and site location. Table D-2 lists the radionuclide-specific constants applied to the intruder scenario calculations.

A detailed discussion of the calculation of doses due to ingrowth of decay products is provided in Appendix E.

References for Appendix D

Baes, C.F. III, R.D. Sharp, A.L. Sjoreen, and R.W. Shor, 1984. *A Review and Analysis of Parameters for Assessing Transport of Environmentally Released Radionuclides through Agriculture.* ORNL-5786. Oak Ridge National Laboratory.

EPA, 1988. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion.* Federal Guidance Report No. 11, Office of Radiation Programs, EPA 520/1-88-020, September.

EPA, 1993. *External Exposure to Radionuclides in Air, Water, and Soil.* Federal Guidance Report No. 12. EPA 402-R-93-081. Oak Ridge National Laboratory and U.S. Environmental Protection Agency.

ICRP (International Commission on Radiological Protection) 1975. International Commission on Radiological Protection, Task Group Report on Reference Man, ICRP Publication No. 23, Pergamon Press, NY.

NRC, 1977. Regulatory Guide 1.109. Calculation of Annual Doses to Man from Routine Release of Reactor Effluents for the Purpose of Evaluating Compliance with 10 CFR Part 50, Appendix I.

WSRC, 1994. *Radiological Performance Assessment for the E-Area Vaults Disposal Facility (U).* WSRC-RP-94-218, Rev. 0. Aiken, SC: EG&G Idaho, Inc., and Westinghouse Savannah River Company.

APPENDIX E:

TREATMENT OF DECAY PRODUCTS IN THE DEVELOPMENT OF PATHWAY AND SCENARIO DOSE CONVERSION FACTORS

NOMENCLATURE

EPA	Environmental Protection Agency
DCF	Dose conversion factor
PDCF	Pathway dose conversion factor
PE	Performance evaluation
SDCF	Scenario dose conversion factor

APPENDIX E:
TREATMENT OF DECAY PRODUCTS IN THE DEVELOPMENT OF PATHWAY AND SCENARIO DOSE CONVERSION FACTORS

E.1 INTRODUCTION

Three transport pathways are addressed in the performance evaluations (PEs): water, atmospheric, and intrusion. The atmospheric pathway involves releases of radionuclides from the waste into the air and exposure through direct inhalation, and ingestion of plants, beef, and milk. The water pathway involves transport through surface or groundwater and exposure from ingestion of drinking water only. The intrusion scenarios involve exposure from direct inhalation of airborne soil particles, ingestion of plants and soil, and external exposure. Fifty-eight radionuclides are evaluated for the water and intrusion pathways. The radionuclides H-3 and C-14 are the ones considered for the atmospheric pathway. To estimate limits for individual radionuclides in the waste, dose conversion factors (DCFs) are necessary to convert doses (expressed as performance objectives) to concentrations. Two types of modified DCFs are used in the PE. The pathway dose conversion factor (PDCF) is the published DCF (dose per unit activity of a radionuclide) (EPA, 1988; EPA, 1993) modified by the intake rates and other applicable exposure pathway parameters for a specific environmental transport pathway. The different types of PDCFs considered in this PE analysis are listed in Table E-1. The second type of modified DCF is the scenario dose conversion factor (SDCF), which is the sum of PDCFs for a specific scenario (e.g., atmospheric transport or intruder scenario). The SDCFs and associated pathways evaluated in the PE are also presented in Table E-1.

Table E-1. PDCFs and SDCFs

	Scenarios	Pathways
PDCFs		water ingestion inhalation vegetation ingestion soil ingestion beef ingestion milk ingestion external exposure
SDCFs	Atmospheric transport	= inhalation PDCF + vegetable ingestion PDCF + beef ingestion PDCF + milk ingestion PDCF
	Intrusion	= inhalation PDCF + vegetable ingestion PDCF + soil ingestion PDCF + external PDCF

The specific equations for calculating water ingestion PDCFs, atmospheric SDCFs, and the intruder SDCFs are presented in Section 5.4.1, Appendix C, and Appendix D, respectively, of this volume. Because some radionuclides have decay products (progeny) that significantly contribute to the dose of a parent radionuclide, the PDCF or SDCF for the parent must include the dose contributions for the decay products. The way in which parents with significant progeny are treated depends on whether the parent and progeny are in secular equilibrium. This appendix presents the different cases of radionuclides and how they are each treated for calculating PDCFs and SDCFs. Because the atmospheric release pathway considered only H-3 and C-14, which do not have progeny, only the water and intrusion pathways are discussed below. The PDCFs for the water pathway and SDCFs for intrusion used in the PE are presented in this appendix; the atmospheric pathway SDCFs are presented in Appendix C.

E.2 CATEGORIZATION OF PARENT RADIONUCLIDES ACCORDING TO THEIR DECAY PRODUCTS

For radionuclides with no radiologically significant decay products, the radionuclide-specific PDCF or SDCF alone is used to calculate the permissible waste concentration. For radionuclides with decay products that could contribute significantly to the dose, the PDCFs/SDCFs depend on whether the decay products are in secular equilibrium at the time of interest, and whether they are long- or short-lived and cause an increase in dose due to ingrowth. The cases of parent/progeny relationships defined in Section 5.4.2 are summarized below in Table E-2. Table E-3 lists the parent radionuclides and significant progeny included in the PE analysis and identifies which progeny fall under which case.

Table E-2. Parent/Progeny Relationships

Case 0	Parent has no progeny that significantly contribute to dose.
Case 1	Progeny reach secular equilibrium with parent before time frame of interest (i.e., before 100 y - time frame of institutional controls).
Case 2	Parent has progeny that do not reach secular equilibrium before the time frame of interest (i.e., before the time of arrival at the groundwater compliance point or the time of intrusion) and are shorter-lived than the parent.
Case 3	Parent has progeny that do not reach secular equilibrium in the time frame of interest and are longer-lived than the parent.

Table E-3. Radioactive Decay Products Considered in the Performance Evaluation (Part 1 of 2)

No.	Isotope	Half-Life (y)	Decay Chains and Radiologically Significant Progeny [a]	Case [b]
1	H-3	1.23E+01	Stable nuclide; No radioactive progeny	0
2	C-14	5.73E+03	Stable nuclide; No radioactive progeny	0
3	Al-26	7.30E+05	Stable nuclide; No radioactive progeny	0
4	Si-32	1.00E+02	P-32 (1.0)	1
5	Cl-36	3.01E+05	Stable nuclide; No radioactive progeny	0
6	K-40	1.28E+09	Stable nuclide; No radioactive progeny	0
7	Co-60	5.27E+00	Stable nuclide; No radioactive progeny	0
8	Ni-59	7.60E+04	Stable nuclide; No radioactive progeny	0
9	Ni-63	1.00E+02	Stable nuclide; No radioactive progeny	0
10	Se-79	6.50E+04	Stable nuclide; No radioactive progeny	0
11	Sr-90	2.91E+01	Y-90 (1.0)	1
12	Zr-93	1.50E+06	Nb-93m (1.0)	1
13	Nb-93m	1.61E+01	Stable nuclide; No radioactive progeny	0
14	Nb-94	2.00E+04	Stable nuclide; No radioactive progeny	0
15	Tc-99	2.13E+05	Stable nuclide; No radioactive progeny	0
16	Pd-107	6.50E+06	Stable nuclide; No radioactive progeny	0
17	Ag-108m	1.30E+02	Ag-108 (0.093)	1
18	Cd-113m	1.41E+01	Stable nuclide; No radioactive progeny	0
19	Sn-121m	5.50E+01	Sn-121 (0.224)	1
20	Sn-126	1.00E+05	Sb-126m (1.0), Sb-126 (0.14)	1
21	I-129	1.57E+07	Stable nuclide; No radioactive progeny	0
22	Cs-135	2.30E+06	Stable nuclide; No radioactive progeny	0
23	Cs-137	3.02E+01	Ba-137m (0.946)	1
24	Ba-133	1.05E+01	Stable nuclide; No radioactive progeny	0
25	Sm-151	9.00E+01	Stable nuclide; No radioactive progeny	0
26	Eu-152	1.35E+01	No significant progeny	0
27	Eu-154	8.59E+00	Stable nuclide; No radioactive progeny	0
28	Pb-210	2.23E+01	Po-210 (1.0)	1
29	Ra-226	1.60E+03	Pb-214 (1.0), Bi-214 (1.0), Pb-210 (1.0), et seq., Po-210 (1.0)	1
30	Ra-228	5.76E+00	Ac-228 (1.0), Th-228 (1.0), Ra-224 (1.0), Pb-212 (1.0), Bi-212 (1.0), Tl-208 (0.3593)	1
31	Th-229	7.30E+03	Ra-225 (1.0), Ac-225 (1.0), Fr-221 (1.0), Bi-213 (1.0), Tl-209 (0.0216)	1
32	Th-230	7.54E+04	Ra-226 (1.0), et seq.	2
33	Th-232	1.40E+10	Ra-228 (1.0), et seq.	1
34	Pa-231	3.28E+04	Ac-227 (1.0), Ra-223 (1.0), Pb-211 (1.0), Bi-211 (1.0), Tl-207 (1.0), Th-227 (0.9862)	1

Table E-3. Radioactive Decay Products Considered in the Performance Evaluation (Part 2 of 2)

No.	Isotope	Half-Life (y)	Decay Chains and Radiologically Significant Progeny [a]	Case [b]
35	U-232	7.00E+01	Th-228 (1.0), Ra-224 (1.0), Pb-212 (1.0), Bi-212 (1.0), Tl-208 (0.3593)	1
36	U-233	1.59E+05	Th-229 (1.0), et seq.	2
37	U-234	2.46E+05	Th-230 (1.0), et seq.	2
38	U-235	7.04E+08	Th-231 (1.0),	1
			Pa-231 (1.0), et seq.	2
39	U-236	2.34E+07	Th-232 (1.0), et seq.	3
40	U-238	4.47E+09	Th-234 (1.0), Pa-234m (1.0), Pa-234 (0.0016)	1
			U-234 (1.0), et seq.	2
41	Np-237	2.14E+06	Pa-233 (1.0)	1
			U-233 (1.0), et seq.	2
42	Pu-238	8.77E+01	U-234 (1.0), et seq.	3
43	Pu-239	2.41E+04	U-235 (1.0), et seq.	3
44	Pu-240	6.56E+03	U-236 (1.0), et seq.	3
45	Pu-241	1.44E+01	Am-241 (1.0), et seq.	3
46	Pu-242	3.75E+05	U-238 (1.0), et seq.	3
47	Pu-244	8.00E+07	Np-240m (1.0)	1
			Pu-240 (1.0), U-236 (1.0)	2
			Th-232 et seq. (1.0)	3
48	Am-241	4.33E+02	Np-237 (1.0), et seq.	3
49	Am-243	7.37E+03	Np-239 (1.0)	1
			Pu-239 (1.0), et seq.	3
50	Cm-243	2.91E+01	Pu-239 (1.0), et seq.	3
51	Cm-244	1.81E+01	Pu-240 (1.0), et seq.	3
52	Cm-245	8.50E+03	Pu-241 (1.0)	1
			Am-241 (1.0), et seq.	2
53	Cm-246	4.76E+03	Pu-242 (1.0), et seq.	3
54	Cm-247	1.56E+07	Pu-243 (1.0)	1
			Am-243 (1.0), et seq.	2
55	Cm-248	3.48E+05	Pu-244 (0.9174), et seq.	3
56	Cf-249	3.51E+02	Cm-245 (1.0), et seq.	3
57	Cf-250	1.31E+01	Cm-246 (1.0), et seq.	3
58	Cf-251	9.00E+02	Cm-247 (1.0), et seq.	3

a Numbers in parentheses indicate branching fractions for progeny nuclides.
Radon is excluded from dose calculations in this Performance Evaluation analysis.
Source: RadDecay Program; version 3.03

b Case 0 No significant radioactive progeny
Case 1 Parent and progeny are in secular equilibrium
Case 2 Parent and progeny are not in secular equilibrium
Case 3 Parent and progeny are not in secular equilibrium; progeny are longer-lived than parent

E.3 DOSE CONVERSION FACTORS FOR THE GROUNDWATER PATHWAY

For the groundwater transport pathway, permissible waste concentrations are calculated at the performance boundary. The times of arrival at the performance boundary may vary for each chemical species. Therefore, the PDCFs must be calculated for the specific time of arrival. A single PDCF is desired for each parent radionuclide so that a single waste concentration limit can be calculated. The way in which the parent PDCFs are calculated depends on the parent/progeny relationship (Cases 0-3). Table E-4 summarizes how each parent/progeny case is treated in the groundwater analysis. The progeny for each parent radionuclide used in the groundwater analysis are listed in Table E-3. Note that some of the progeny listed in Table E-3 are not relevant for the water ingestion pathway (e.g., they only contribute an external dose).

Table E-4. Method of Groundwater Analysis

Case	Method of Groundwater Analysis	Nuclides in PE
0	Transport parent to compliance point and compute dose.	H-3, C-14, Al-26, Cl-36, K-40, Co-60, Ni-59, Ni-63, Se-79, Nb-93m, Nb-94, Tc-99, Pd-107, Cd-113m, I-129, Cs-135, Ba-133, Sm-151, Eu-152, Eu-154 *Total: 20 nuclides*
1	Transport parent to compliance point, assume all progeny are produced at that time and place, and compute dose from the summed contribution of parent plus progeny (accounting for branching fractions of progeny).	Si-32, Sr-90, Zr-93, Ag-108m, Sn-121m, Sn-126, Cs-137, Pb-210, Ra-226, Ra-228, Th-229, Th-232, Pa-231, U-232, Np-237 *Total: 15 nuclides*
2	Transport parent to compliance point, then use Bateman equations to predict activities of parent and progeny at time of arrival; use activities as multipliers for parent and progeny dose conversion factors to compute combined dose.	Th-230, U-233, U-234, U-235, U-238, Pu-244, Cm-245, Cm-247 *Total: 8 nuclides*
3	Compute dose as the greater of: (a) Parent (and/or any short-lived progeny in secular equilibrium) transported by itself, i.e., same as Case 1; (b) Progeny transported by itself, then compute dose on a "per curie of parent" basis by reducing the progeny dose by the ratio of half-lives.	Pu-238, Pu-241, Am-241, Cm-243, Cu-244, Cf-249, Cf-250 *Total: 7 nuclides*
	Ignore decay products due to negligible ingrowth during the compliance period.	U-236, Pu-239, Pu-240, Pu-242, Am-243, Cm-246, Cm-248, Cf-251 *Total: 8 nuclides*

Case 0 can be defined as those parents with no decay products. Case 1 is for parents with decay products that are always assumed to be in secular equilibrium. For both Case 0 and Case 1, a single pathway dose conversion factor (PDCF) value can be readily developed for each parent nuclide. These PDCFs have been computed based upon the consumption by an adult of an average of 2 L/day of drinking water.

Case 2 and 3 radionuclides require special attention for the groundwater pathway. Case 2 is where the decay products are shorter-lived than the parents, but they do not reach equilibrium with their parents within the 10,000-y compliance period. For Case 2 parents, Bateman equations are to be used to determine the contributions of the radiological significant decay products to the overall PDCF at the time of arrival at the performance boundary.

Only nine radionuclides need to be examined initially under Case 2: Th-230, U-233, U-234, U-235, U-238, Np-237, Pu-244, Cm-245, and Cm-247. However, the Case 2 decay products of Np-237 can be ignored during the 10,000-y compliance period due to insufficient ingrowth. Thus, only eight radionuclides are of interest under Case 2, for developing PDCF values.

Table E-5 presents the eight Case 2 radionuclides and the ratios of the maximum and minimum PDCFs during the compliance period. Data in the table indicate that the ratio of maximum to minimum PDCFs during the compliance period for U-238, Pu-244, Cm-245, and Cm-247 does not vary by a factor of more than 1.8. Therefore, for these four nuclides, a single PDCF value has been developed based upon the maximum PDCF during the compliance period. This single PDCF value is used in the groundwater analysis, regardless of the time of arrival at the performance boundary.

Table E-5. Summary of Dose Conversion Factors for Case 2 Parent Radionuclides for the Groundwater Pathway

Nuclide	Arrival Time at Performance Boundary (y)		Ratio of Pathway Dose Conversion Factors (PDCFs) Max : Min	Maximum PDCF (rem/y per µCi/L)
	For Maximum Dose	For Minimum Dose		
Th-230	9,000	100	10.3	6140
U-233	10,000	100	8.2	1953
U-234	10,000	100	3.1	648
U-235	10,000	100	16.4	3725
U-238	10,000	100	1.0	207
Pu-244	10,000	100	1.7	4112
Cm-245	1,000	10,000	1.8	4565
Cm-247	10,000	100	1.7	4342

Table E-5 also shows that Th-230, U-233, U-234, and U-235 have PDCF values that increase throughout the compliance period by factors of up to 16. A set of curve fits were developed for these four radionuclides. The equations developed for these four radionuclides are valid only over the period 100 to 10,000 y after closure of the facility. In the event that arrival times at the compliance point exceed 10,000 y, the equations will not produce valid PDCF values. In that case, the recommended approach is to use the PDCF value at 10,000 y instead of the equation. Note that subsurface travel times in excess of 10,000 y are used to disqualify the groundwater pathway from determining the most restrictive permissible waste concentrations.

For all Case 2 radionuclides, the total PDCF values are already "decayed" (i.e., via use of Bateman equations); hence, the decay factor (r_{Decay}) should not be used as part of the equation for dose from consumption of contaminated groundwater.

For Case 3, the decay products are longer-lived than their parents. Case 3 doses must be computed as the greater of (1) the parent transported to the point of compliance,and (2) the longer-lived decay products transported to the point of compliance and with their dose reduced by the ratio of the half-lives of decay products to the parent.

Among the Case 3 radionuclides, the decay product doses can be ignored (due to negligible ingrowth during the compliance period) for the following parent radionuclides: U-236, Pu-239, Pu-240, Pu-242, Am-243, Cm-246, Cm-248, and Cf-251. For these parents, the appropriate PDCFs are developed in the same manner as for Case 0 or Case 1, whichever is appropriate for the specific parent.

For all other Case 3 parents (listed in Table E-4), the "ratio of half-lives" rule can be applied as necessary. When a decay product is longer-lived than its parent, the maximum activity of the decay product is approximately equal to the initial activity of the parent multiplied by the ratio of the half-lives of the decay product and the parent, but there is never any kind of equilibrium between the activities of the parent and decay product. If exposures are expected to occur at a time sufficiently long after disposal such that the activity of the shorter-lived parent is reduced to innocuous levels, a simple and conservative approach would be to convert the initial activity of the parent to an equivalent initial activity of the decay product using the ratio of the two half-lives (Equation E-1). Then the activity of the decay product is used as input to the transport and exposure analysis.

$$A_d = A_p * (t_{1/2,d} \ / \ t_{1/2,p}) \qquad \text{(E-1)}$$

where

A_d = initial activity of the decay product
A_p = initial activity of the parent
$t_{1/2,d}$ = half-life of the decay product
$t_{1/2,p}$ = half-life of the parent

Table E-6 presents water pathway PDCF values for all 58 radionuclides in the PEs. For four radionuclides (Th-230, U-233, U-234, and U-235) equations have been developed to relate the PDCF to time of arrival at the performance boundary during the 10,000-y compliance period;

however, the curve fit equations are valid only until 10,000 y. Beyond 10,000 y, the maximum PDCF value should be used.

E.4 DOSE CONVERSION FACTORS FOR INTRUSION SCENARIOS

The radionuclide-specific methodology for calculating intruder SDCFs is summarized in Table E-7. For the generic intruder analysis, the times of intrusion are generally limited to one of the following, depending on the disposal technology and intruder scenario: 100, 300, or 500 y. Case 1 is when parents and progeny reach secular equilibrium before the time of intrusion. In this case, the SDCF is the sum of the SDCFs for the parent and the significant progeny, weighted by any applicable branching fractions for the progeny.

In Cases 2 and 3, the SDCFs for parents and progeny, which do not reach secular equilibrium by the time of intrusion, are calculated for various times up to 10,000 y (the compliance period in this PE analysis). For the Case 2 parents, the decay products cause an increase in dose over time. For these radionuclides, the time of intrusion was assumed to be at the time of maximum dose. For six radionuclides (U-233, U-234, U-235, U-238, Pu-244, and Cu-247) this was 10,000 y. For two radionuclides the time of maximum dose occurs before the end of the 10,000-y compliance period. The times of intrusion for these were as follows: Th-230 - 9,000 y and Cm-235 - 1,000 y. The SDCFs are then calculated as the weighted sum of the parent and the decay product, based on the activity fraction at that time of maximum dose (or 10,000 y, which ever comes first). For the Case 3 parents, the progeny are not in equilibrium within the time frame of interest, but the doses do not increase over time. In this case, the dose is taken at the specific intrusion time of interest (e.g., 100, 300, or 500 y) with the appropriate activity fractions for that time. Some of the Case 3 radionuclides have progeny that are not in secular equilibrium with the parent, but they are ignored due to negligible ingrowth during the compliance period. These are treated as if they have no progeny (i.e., equivalent to a Case 0).

Table E-8 displays the intrusion SDCFs used in the PE. The SDCFs for the trench and tumulus are the same for both intruder scenarios. The only differences in the two technologies for the intrusion scenarios are the time of intrusion and the geometry factor. These two parameters are used when calculating the final waste concentration limit (described in Sect. 5.3.3.2 and Appendix D).

For the Case 2 and 3 nuclides, the SDCFs for the progeny (not in secular equilibrium) are listed separately in Table E-8. The final SDCF for the parent is calculated using the activity fractions of parent to progeny at the intrusion time of interest.

References for Appendix E

EPA, 1988. *Limiting Values of Radionuclide Intake and Air Concentration and Dose Conversion Factors for Inhalation, Submersion, and Ingestion.* Federal Guidance Report No. 11. EPA 520/1-88-020. Office of Radiation Programs.

EPA, 1993. *External Exposure to Radionuclides in Air, Water, and Soil.* Federal Guidance Report No. 12. EPA 402-R-93-081. Oak Ridge National Laboratory and U. S. Environmental Protection Agency.

Table E-6. Water Pathway Dose Conversion Factors (PDCFs) (Part 1 of 2)

No.	Nuclide [1]	Ingestion PDCF [2] rem/y per µCi/L	Remarks [3,4]
1	H-3	4.67E-02	
2	C-14	1.52E+00	
3	Al-26	1.06E+01	
4	Si-32 + d	7.99E+00	
	Si-32		
	P-32		
5	Cl-36	2.21E+00	
6	K-40	1.36E+01	
7	Co-60	1.97E+01	
8	Ni-59	1.53E-01	
9	Ni-63	4.21E-01	
10	Se-79	6.35E+00	
11	Sr-90 + d	1.12E+02	
	Sr-90		
	Y-90		
12	Zr-93 + d	1.59E+00	
	Zr-93		
	Nb-93m		
13	Nb-93m	3.81E-01	
14	Nb-94	5.21E+00	
15	Tc-99	1.07E+00	
16	Pd-107	1.09E-01	
17	Ag-108m	5.56E+00	
18	Cd-113m	1.17E+02	
19	Sn-121m + d	1.28E+00	
	Sn-121m		
	Sn-121		
20	Sn-126 + d	1.53E+01	
	Sn-126		
	Sb-126		
21	I-129	2.01E+02	
22	Cs-135	5.16E+00	
23	Cs-137	3.65E+01	
24	Ba-133	2.48E+00	
25	Sm-151	2.84E-01	
26	Eu-152	4.73E+00	
27	Eu-154	6.97E+00	
28	Pb-210 + d	5.30E+03	
	Pb-210		
	Po-210		
29	Ra-226 + d	6.27E+03	
	Ra-226		
	Pb-210		
	Po-210		
30	Ra-228 + d	1.64E+03	
	Ra-228		
	Th-228		
	Ra-224		
	Pb-212		
31	Th-229 + d	2.94E+03	
	Th-229		
	Ra-225		
	Ac-225		

Table E-6. Water Pathway Dose Conversion Factors (PDCFs) (Part 2 of 2)

No.	Nuclide [1]	Ingestion PDCF [2] rem/y per µCi/L	Remarks [3,4]
32	Th-230	343.6 + 2.644*T - 4.9E-4*T^2 + 4.274E-8*T^3 - 1.44E-12*T^4	Case 2; Curve fit equation; T = time of arrival, after facility closure
33	Th-232 + d	3.63E+03	
	Th-232		
	Ra-228		
	Th-228		
	Ra-224		
	Pb-212		
34	Pa-231 + d	1.85E+04	
	Pa-231		
	Ac-227		
	Ra-223		
35	U-232 + d	1.55E+03	
	U-232		
	Th-228		
	Ra-224		
	Pb-212		
36	U-233	217.3 + 0.26013*T - 8.656E-6*T^2	Case 2; Curve fit equation; T = time of arrival, after facility closure
37	U-234	206.2 + 5.612E-3*T + 8.032E-6*T^2 - 4.175E-10*T^3	Case 2; Curve fit equation; T = time of arrival, after facility closure
38	U-235	188.6 + 0.3909*T - 3.728E-6*T^2	Case 2; Curve fit equation; T = time of arrival, after facility closure
39	U-236	1.96E+02	Case 3; PDCF for parent only
40	U-238	2.07E+02	Case 2; calculated using Bateman equations
41	Np-237	3.24E+03	
42	Pu-238	2.34E+03	Case 3; PDCF for parent only
43	Pu-239	2.58E+03	Case 3; PDCF for parent only
44	Pu-240	2.58E+03	Case 3; PDCF for parent only
45	Pu-241	5.00E+01	Case 3; PDCF for parent only
46	Pu-242	2.45E+03	Case 3; PDCF for parent only
47	Pu-244	4.11E+03	Case 2; calculated using Bateman equations
48	Am-241	2.66E+03	Case 3; PDCF for parent only
49	Am-243	2.64E+03	Case 3; PDCF for parent only
50	Cm-243	1.83E+03	Case 3; PDCF for parent only
51	Cm-244	1.47E+03	Case 3; PDCF for parent only
52	Cm-245	4.57E+03	Case 2; calculated using Bateman equations
53	Cm-246	2.70E+03	Case 3; PDCF for parent only
54	Cm-247	4.34E+03	Case 2; calculated using Bateman equations
55	Cm-248	9.94E+03	Case 3; PDCF for parent only
56	Cf-249	3.46E+03	Case 3; PDCF for parent only
57	Cf-250	1.56E+03	Case 3; PDCF for parent only
58	Cf-251	3.54E+03	Case 3; PDCF for parent only

1 An entry with "+d" denotes parent radionuclide plus its radiologically significant, short-lived decay products that are assumed to be in secular equilibrium with the parent.

2 PDCFs are based on ingestion dose conversion factors (DCFs) in EPA, 1988 and a water consumption rate of 730 L/yr. For radionuclides with more than one DCF, the one with the greatest fractional uptake from the small intestine to the bloodstream was used. These PDCFs do not account for decay.

3 Case 2 indicates progeny and parent that do not reach secular equilibrium within 10,000 years. Bateman equations have been used to compute contributions of progeny to total PDCF. These PDCFs account for decay.

4 The PDCFs for Case 3 are only for the parent and do not account for decay. The total PDCF depends on the activity fraction of the parent and progeny at the arrival time.

Table E-7. Methodology for Calculating Intruder SDCFs Based on Parent/Progeny Relationship

Case	Method of Intruder Analysis	Nuclides in PE
0	The SDCF equals that for the individual parent.	H-3, C-14, Al-26, Cl-36, K-40, Co-60, Ni-59, Ni-63, Se-79, Nb-93m, Nb-94, Tc-99, Pd-107, Cd-113m, I-129, Cs-135, Ba-133, Sm-151, Eu-152, Eu-154 *Total: 20 nuclides*
1	The SDCF equals the weighted sum of SDCFs for the parent and significant progeny. The weighting is based on the branching fraction for each nuclide.	Si-32, Sr-90, Zr-93, Ag-108m, Sn-121m, Sn-126, Cs-137, Pb-210, Ra-226, Ra-228, Th-229, Th-232, Pa-231, U-232, Np-237 *Total: 15 nuclides*
2	(a) SDCF increases over time, but the maximum dose occurs before the end of the compliance period (time of intrusion for Th-230: 9,000 y; Cm-245: 1,000 y). The SDCF equals the weighted sum of SDCFs for the parent and significant progeny. The weighting is based on the activity fractions of the parent and progeny at the time of intrusion.	Th-230, Cm-245 *Total: 2 nuclides*
	(b) SDCF increases over entire compliance period; take the maximum dose at end of the compliance period (i.e., 10,000 y). The SDCF equals the weighted sum of SDCFs for the parent and significant progeny. The weighting is based on the activity fractions of the parent and progeny at the time of intrusion.	U-233, U-234, U-235, U-238, Pu-244, Cm-247 *Total: 6 nuclides*
3	(a) SDCF does not increase with time. The SDCF equals the weighted sum of SDCFs for the parent and significant progeny. The weighting is based on the activity fractions of the parent and progeny at the time of intrusion.	Pu-238, Pu-241, Am-241, Cm-243, Cm-244, Cf-249, Cf-250, Cf-251 *Total: 8 nuclides*
	(b) Progeny not in secular equilibrium can be ignored due to negligible ingrowth during compliance period; SDCF is calculated as if there are no progeny (Case 0).	U-236, Pu-239, Pu-240, Pu-242, Am-243, Cm-246, Cm-248 *Total: 7 nuclides*

Table E-8. Intrusion Scenario Dose Conversion Factors (SDCFs) (Part 1 of 3)

	Nuclide [a]	Homesteader Scenario SDCF (rem/y per µCi/m^3)	Post-Drilling Scenario SDCF (rem/y per µCi/m^3)
1	H-3	3.95E-06	3.95E-07
2	C-14	1.50E-05	1.50E-06
3	Al-26	3.83E-03	1.82E-06
4	Si-32 + d	1.71E-04	1.71E-05
5	Cl-36	1.17E-03	1.17E-04
6	K-40	2.85E-04	5.76E-06
7	Co-60	3.57E-03	1.81E-06
8	Ni-59	7.07E-08	7.07E-09
9	Ni-63	1.95E-07	1.95E-08
10	Se-79	1.25E-06	1.25E-07
11	Sr-90 + d	1.98E-04	1.98E-05
12	Zr-93 + d	4.09E-08	3.09E-09
13	Nb-93m	1.80E-08	1.72E-09
14	Nb-94	2.13E-03	1.08E-06
15	Tc-99	1.21E-05	1.21E-06
16	Pd-107	3.42E-08	3.39E-09
17	Ag-108m + d	2.13E-03	1.50E-06
18	Cd-113m	1.34E-04	1.34E-05
19	Sn-121m + d	5.09E-07	6.99E-09
20	Sn-126 + d	2.61E-03	1.42E-06
21	I-129	8.06E-05	7.78E-06
22	Cs-135	1.21E-06	1.21E-07
23	Cs-137 + d	7.59E-04	1.23E-06
24	Ba-133	4.36E-04	2.61E-07
25	Sm-151	1.15E-08	1.07E-09
26	Eu-152	1.54E-03	7.70E-07
27	Eu-154	1.69E-03	8.48E-07
28	Pb-210 + d	3.10E-04	3.10E-05
29	Ra-226 + d	2.78E-03	3.39E-05
30	Ra-228 + d	3.58E-03	4.63E-06
31	Th-229 + d	4.28E-04	3.02E-06
32	Th-230	1.09E-05	3.45E-07
	Ra-226	1.82E-05	1.80E-06
	Pb-214	2.95E-04	1.60E-07
	Bi-214	2.16E-03	1.02E-06
	Pb-210	2.96E-04	2.95E-05
	Po-210	1.45E-05	1.43E-06
33	Th-232 + d	3.63E-03	6.33E-06
34	Pa-231 + d	8.31E-04	1.92E-05
35	U-232 + d	2.30E-03	5.70E-06
36	U-233	1.19E-05	8.07E-07
	Th-229	1.42E-04	2.27E-06
	Ra-225	7.65E-06	5.24E-07
	Ac-225	1.48E-05	8.80E-08
	Fr-221	3.38E-05	1.85E-08
	Bi-213	1.68E-04	8.76E-08
	Tl-209	2.84E-03	1.35E-06

Table E-8. Intrusion Scenario Dose Conversion Factors (SDCFs) (Part 2 of 3)

	Nuclide [a]	Homesteader Scenario SDCF (rem/y per µCi/m³)	Post-Drilling Scenario SDCF (rem/y per µCi/m³)
37	U-234	1.17E-05	7.92E-07
	Th-230	1.09E-05	3.45E-07
	Ra-226	1.82E-05	1.80E-06
	Pb-214	2.95E-04	1.60E-07
	Bi-214	2.16E-03	1.02E-06
	Pb-210	2.96E-04	2.95E-05
	Po-210	1.45E-05	1.43E-06
38	U-235	1.70E-04	8.31E-07
	Th-231	8.02E-06	4.53E-09
	Pa-231	1.51E-04	7.23E-06
	Ac-227	3.01E-04	1.09E-05
	Th-227	1.15E-04	6.19E-08
	Ra-223	1.42E-04	9.67E-07
	Pb-211	6.74E-05	3.41E-08
	Bi-211	5.63E-05	2.99E-08
	Tl-207	4.36E-06	2.21E-09
39	U-236	1.11E-05	7.50E-07
40	U-238	1.05E-05	7.11E-07
	Th-234	5.38E-06	1.09E-08
	Pa-234m	1.97E-05	9.81E-09
	Pa-234	2.54E-03	1.26E-06
	U-234	1.17E-05	7.92E-07
	Th-230	1.09E-05	3.45E-07
	Ra-226	1.82E-05	1.80E-06
	Pb-214	2.95E-04	1.60E-07
	Bi-214	2.16E-03	1.02E-06
	Pb-210	2.96E-04	2.95E-05
	Po-210	1.45E-05	1.43E-06
41	Np-237 + d	5.27E-04	2.71E-05
42	Pu-238	2.94E-05	1.82E-06
	U-234	1.17E-05	7.92E-07
	Th-230	1.09E-05	3.45E-07
	Ra-226	1.82E-05	1.80E-06
	Pb-214	2.95E-04	1.60E-07
	Bi-214	2.16E-03	1.02E-06
	Pb-210	2.96E-04	2.95E-05
	Po-210	1.45E-05	1.43E-06
43	Pu-239	3.23E-05	2.01E-06
44	Pu-240	3.23E-05	2.01E-06
45	Pu-241	6.24E-07	3.88E-08
	Am-241	4.71E-05	2.48E-06
	Np-237	3.02E-04	2.70E-05
	Pa-233	2.24E-04	1.21E-07
	U-233	1.19E-05	8.07E-07
	Th-229	1.42E-04	2.27E-06
	Ra-225	7.65E-06	5.24E-07
	Ac-225	1.48E-05	8.80E-08
	Fr-221	3.38E-05	1.85E-08
	Bi-213	1.68E-04	8.76E-08
	Tl-209	2.84E-03	1.35E-06

Table E-8. Intrusion Scenario Dose Conversion Factors (SDCFs) (Part 3 of 3)

	Nuclide [a]	Homesteader Scenario SDCF (rem/y per µCi/m³)	Post-Drilling Scenario SDCF (rem/y per µCi/m³)
46	Pu-242	3.08E-05	1.91E-06
47	Pu-244	3.04E-05	1.88E-06
	Np-240m	4.44E-04	2.22E-07
	Pu-240	3.23E-05	2.01E-06
48	Am-241	4.71E-05	2.48E-06
	Np-237	3.02E-04	2.70E-05
	Pa-233	2.24E-04	1.21E-07
	U-233	1.19E-05	8.07E-07
	Th-229	1.42E-04	2.27E-06
	Ra-225	7.65E-06	5.24E-07
	Ac-225	1.48E-05	8.80E-08
	Fr-221	3.38E-05	1.85E-08
	Bi-213	1.68E-04	8.76E-08
	Tl-209	2.84E-03	1.35E-06
49	Am-243	6.85E-05	2.48E-06
	Np-239	1.66E-04	9.11E-08
	Pu-239	3.23E-05	2.01E-06
50	Cm-243	1.51E-04	1.45E-06
	Pu-239	3.23E-05	2.01E-06
51	Cm-244	1.82E-05	1.11E-06
	Pu-240	3.23E-05	2.01E-06
52	Cm-245	1.08E-04	2.10E-06
	Pu-241	6.24E-07	3.88E-08
	Am-241	4.71E-05	2.48E-06
53	Cm-246	3.33E-05	2.04E-06
54	Cm-247	4.22E-04	2.09E-06
	Pu-243	1.75E-05	9.81E-09
	Am-243	6.85E-05	2.48E-06
	Np-239	1.66E-04	9.11E-08
	Pu-239	3.23E-05	2.01E-06
55	Cm-248	1.22E-04	7.50E-06
56	Cf-249	4.50E-04	2.82E-06
	Cm-245	1.08E-04	2.10E-06
	Pu-241	6.24E-07	3.88E-08
	Am-241	4.71E-05	2.48E-06
57	Cf-250	1.92E-05	1.17E-06
	Cm-246	3.33E-05	2.04E-06
	Pu-242	3.08E-05	1.91E-06
58	Cf-251	1.59E-04	2.73E-06
	Cm-247	4.22E-04	2.09E-06
	Pu-243	1.75E-05	9.81E-09
	Am-243	6.85E-05	2.48E-06
	Np-239	1.66E-04	9.11E-08
	Pu-239	3.23E-05	2.01E-06

a "+d" Indicates that decay products are included and are assumed to be in secular equilibrium with the parent.

Indented radionuclides are decay products not in secular equilibrium with the parent. The total SDCFs for the parents are the sum of the SDCFs for the parent and progeny, weighted by the activity fraction of the parent and progeny at the intrusion time of interest.

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