# Granite Disposal of U.S. High-Level Radioactive Waste

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## Abstract

This report evaluates the feasibility of disposing U.S. high-level radioactive waste in granite several hundred meters below the surface of the earth. The U.S. has many granite formations with positive attributes for permanent disposal. Similar crystalline formations have been extensively studied by international programs, two of which, in Sweden and Finland, are the host rocks of submitted or imminent repository license applications. This report is enabled by the advanced work of the international community to establish functional and operational requirements for disposal of a range of waste forms in granite media. In this report we develop scoping performance analyses, based on the applicable features, events, and processes (FEPs) identified by international investigators, to support generic conclusions regarding post-closure safety.

Unlike the safety analyses for disposal in salt, shale/clay, or deep boreholes, the safety analysis for a mined granite repository depends largely on waste package preservation. In crystalline rock, waste packages are preserved by the high mechanical stability of the excavations, the diffusive barrier of the buffer, and favorable chemical conditions. The buffer is preserved by low groundwater fluxes, favorable chemical conditions, backfill, and the rigid confines of the host rock. An added advantage of a mined granite repository is that waste packages would be fairly easy to retrieve, should retrievability be an important objective.

The results of the safety analyses performed in this study are consistent with the results of comprehensive safety assessments performed for sites in Sweden, Finland, and Canada. They indicate that a granite repository would satisfy established safety criteria and suggest that a small number of FEPs would largely control the release and transport of radionuclides. In the event the U.S. decides to pursue a potential repository in granite, a detailed evaluation of these FEPs would be needed to inform site selection and safety assessment.

# ACKNOWLEDGEMENTS

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# ACRONYMS

| | |
|---|---|
| BWR | Boiling water reactor |
| DOE | U.S. Department of Energy |
| EBS | Engineered barrier system |
| EDZ | Excavation damage zone |
| EPA | U.S. Environmental Protection Agency |
| ERB1B | Example Reference Biosphere 1B |
| FEPs | Features, events, and processes |
| HLW | High-level radioactive waste |
| IAEA | International Atomic Energy Agency |
| ICRP | International Commission on Radiation Protection |
| MOX | Mixed oxide (fuel) |
| NM | New Mexico |
| NRC | U.S. Nuclear Regulatory Commission |
| NTS | Nevada Test Site |
| NWPA | Nuclear Waste Policy Act (1984) |
| PA | Performance assessment |
| PWR | Pressurized water reactor |
| RMEI | Reasonably maximally exposed individual |
| RW | Radioactive waste |
| SKB | Svensk Kärnbränslehantering AB |
| SNF | Spent nuclear fuel |
| TRU | Transuranic (waste) |
| UNF | Used nuclear fuel |
| UOX | Uranium oxide (fuel) |
| URL | Underground research laboratory |
| WP | Waste package |
| WIPP | Waste Isolation Pilot Plant |

# 1. INTRODUCTION

This report analyzes the potential technical features and projected safety performance of a repository for high-level radioactive waste (HLW) in suitable granite formations in the United States. HLW includes commercial used nuclear fuel (UNF) (also called spent nuclear fuel or SNF), existing HLW of the U.S. Department of Energy (DOE), and HLW from the reprocessing of commercial UNF. Because U.S. efforts have focused on the volcanic tuff site at Yucca Mountain, radioactive waste disposal in U.S. granite formations has not been considered for many years. However, several countries have actively studied nuclear waste disposal in granite media for decades. The analysis reported here, although generic, draws heavily on extensive information from international repository programs that have continued to advance repository concepts in granite (Table 1-1).

Table 1-1. Granite HLW Repository Programs around the World (Hansen et al. 2011).

| Country | Comments |
|---------|----------|
| Canada | Restarting the repository siting program; hosted multi-national research on granite in the 1980s and 1990s in the now closed Pinawa Underground Research Laboratory (URL). |
| Finland | Scheduled to open the nation's first HLW repository in granite in 2020 in the voluntary host municipality of Eurajoki. Opened the nearby Onkalo URL in granite in 2010. |
| Japan | Pursuing voluntary candidate repository sites with an expressed interest in granite. Operates URLs in clay and granite. |
| Sweden | Scheduled to open the nation's first repository for HLW in granite at Forsmark in the voluntary host community of Oesthammar in 2025. Operates a multi-national URL in granite at Äspö. |
| United States of America | Has operated URLs in basalt, granite, salt, and tuff. |

Various references to lithology and types of geologic formations are used in this report. Potentially suitable crystalline rock formations share many characteristics favorable to repository development and waste isolation. The lithologies encompass a broad range of material properties. For this report we use the term "granite" broadly to represent the full range of potentially suitable crystalline rock formations as opposed to the strict geologic definition of the lithology.

The remainder of this introductory section briefly reviews the history of granite repository research in the U.S. and around the world (Section 1.1), the U.S. legal and regulatory framework (Section 1.2), and available U.S. granite formations (Section 1.3). The other major sections of this report describe the generic technical and performance analyses for disposal of U.S. HLW and spent nuclear fuel (SNF) in granite media. Section 2 outlines the technical basis for disposal in granite, including descriptions of mature granite repository concepts and expected evolution of

the repository and host rock over time. Section 3 considers potential release scenarios and the framework for a performance assessment (PA) for a granite repository. Section 4 describes the generic performance analysis developed in this report. Section 5 concludes with a summary and recommendations for future work.

## 1.1. Brief History of Granite Repository Research

Research in underground disposal of radioactive waste in the U.S. began in earnest in the 1950s. The National Academy of Sciences released a report in 1957 concluding that disposal of radioactive waste in salt beds or salt domes may offer "the most practical immediate solution of the problem" (Hess et al. 1957). This seminal report prompted U.S. research to focus largely on salt formations in the early years. At the same time, countries without salt formations focused on repository concepts in other media such as granite.

By the late 1970s and early 1980s, U.S. repository research in granite formations had broadened considerably. The U.S. developed a URL at a depth of 420 m in the Climax monzonite stock, a granitic body at the Nevada Test Site (NTS) (Figure 1-1). Work at this URL demonstrated the feasibility and safety of spent fuel storage and retrieval from an underground facility in granite (Patrick 1986).



Figure 1-1.  Waste Disposal Demonstration in Granite at the Nevada Test Site (Patrick 1986).

The Nuclear Waste Policy Act (NWPA) of 1982 required the federal government to take responsibility for the disposal of civilian SNF and HLW. As part of this process, the NWPA directed the government to identify three potential repository sites and established a timetable leading to the opening of the first repository. In succeeding years a screening process was conducted to identify potentially viable sites. Granitic sites were considered initially, but were not carried forward into the screening process.

U.S. research on disposal in crystalline rock ended soon after the Nuclear Waste Policy Amendments Act of 1987 was enacted. This act directed the Secretary of Energy to characterize only the Yucca Mountain site and to phase out funding for all research programs designed to evaluate the suitability of crystalline rock as a potential repository host medium.

Although crystalline rock was no longer considered to be a potential repository host rock, limited work on crystalline concepts continued in the U.S. as part of other research. In the early 1990s, Sandia National Laboratories developed a probability-based PA model for disposal of SNF in granite to compare to a similar model for salt. Multiple codes and algorithms, simulating processes ranging from corrosion to the flow and transport of fluids, energy, and radionuclides through the repository and fractured granite, were linked and executed in a set of Monte Carlo simulations. The results indicated that, with adequate treatment and packaging, all the waste forms considered could be isolated from the environment with acceptable performance for 10,000 years (SNL 1993). For that study, the waste package outer layer was assumed to be Inconel 625 (69% Ni, 22% Cr, 9% Mo) rather than copper, the preferred material in current granite repository concepts in Sweden, Finland, and Canada (SKB 2011; Posiva 2010; Garisto et al. 2010).

In 2008, the U.S. DOE submitted a license application to the U.S. Nuclear Regulatory Commission (NRC) for a proposed repository at Yucca Mountain, Nevada. Two years later, at the direction of the Obama administration, the U.S. DOE filed a motion to withdraw the application. As a result, other disposal environments in the U.S. are once again being investigated, including disposal in granite.

Granitic repository programs in Europe, North America, and Asia, which have matured over the past three decades and have generated large sets of data from laboratory experiments and *in situ* tests performed in URLs, provide a valuable resource for this effort. These programs have produced corresponding refinements to repository design concepts and medium-specific performance analyses. Two of these programs have either entered the license application phase (Sweden) or are very close to doing so (Finland). For the generic evaluation presented in this report, design and operational concepts from these international programs are used as the technical basis. This use of existing work permits a credible analysis that applies to the many granite formations in the U.S. that have the potential to host a high level radioactive waste repository.

## 1.2. Legal and Regulatory Framework

The 1987 Amendments to the NWPA restrict the pursuit of geologic repositories in the U.S. to a single site in volcanic tuff at Yucca Mountain in Nevada. Hence, at a minimum, pursuit of a HLW/UNF repository in other formations would require changes to the NWPA.

In principle, the existing U.S. Environmental Protection Agency (EPA) HLW/UNF radiation protection standards and the NRC HLW/UNF regulatory framework originally promulgated in the 1980s (40 CFR 191 and 10 CFR 60, respectively), which predate the selection of Yucca Mountain, could apply to a repository in another geologic medium. Indeed, a specific derivative (40 CFR Part 194) of the EPA standard, 40 CFR 191, currently applies to the Waste Isolation Pilot Plant (WIPP), a repository located in New Mexico and situated in bedded salt for disposal of transuranic (TRU) waste from defense activities. In 40 CFR 191, the primary indicator of risk to human health is the cumulative release of radionuclides. Its measure is the complementary cumulative distribution function of the cumulative mass release of radionuclides that cross a boundary 5 km from the site $10^4$ years after disposal, normalized by (a) EPA-derived limits for specified radionuclides and (b) the mass of radionuclides placed in the repository. However, in 1995 the National Research Council of the National Academies of Science and Engineering recommended using dose as the primary indicator of risk to human health for a Yucca Mountain repository. The International Commission on Radiation Protection (ICRP) made a similar recommendation in 1997 (ICRP 1997), and the International Atomic Energy Agency (IAEA) model standard, issued in 2006, uses a dose indicator for deep geologic disposal of radioactive waste (IAEA 2006). Because of these changes, this analysis assumes dose is the primary hazard indicator for radioactive waste disposal in granite.

The EPA standard, 40 CFR 197, specifically written for a repository at Yucca Mountain, specifies the hazard indicator measure as the expected (mean) peak dose to a reasonably maximally exposed individual (RMEI) living along the predominant groundwater flow path 18 km from the site. The standard set a limit on expected peak dose rate of 0.15 mSV yr$^{-1}$ (15 mrem yr$^{-1}$) before $10^4$ years and 1 mSv yr$^{-1}$ (100 mrem yr$^{-1}$) between $10^4$ and $10^6$ years. The latter limit is consistent with the ICRP and IAEA recommendations. The characteristics of the hypothetically exposed individual are those of the RMEI defined in 40 CFR 197. These characteristics are appropriate for humans living in arid regions similar to Yucca Mountain but may need to be reconsidered for granite disposal sites in less arid regions. For the performance analysis presented in Section 4, the same dose limits were applied but the distance from the repository to the RMEI was shortened to 500 m to be consistent with the pre-existing flow model adopted for the analysis. The EPA and NRC regulations pertaining to HLW disposal place specific requirements on PA models that are intended to demonstrate compliance with regulatory performance objectives. The generic performance analysis presented in Section 4 of this report represents what are likely to be the major features of a compliant PA, but the analysis itself does not meet all the regulatory requirements for a PA.

Other details of the regulatory framework, including screening criteria for potentially relevant FEPs and guidance on inadvertent human intrusion, are assumed to be unchanged from those stated in 40 CFR 197.

## 1.3. Granites in the United States

As a class of rocks, granite is a crystalline igneous rock that encompasses a variety of specific lithologies. Formed from magma that intrudes other rocks deep within the continental crust, it predominantly consists of quartz and feldspar. Major granite formations in the U.S. are summarized in Table 1-2. Granite outcrops are mapped in Figure 1-2.

Table 1-2.  Granite Bodies in the United States.

| Location | Units | Attributes | Comments |
|---|---|---|---|
| Pacific Border Region | Triassic, Jurassic, and Cretaceous granites, California, Oregon, and Washington | Very high hydraulic gradient. Large vertical movement along the Sierra Nevada and Cascade Range. Large horizontal movement along the coast range. High seismic and volcanic activity. Regions of large expanses of granite. | |
| Basin and Range Region | Precambrian granites, Arizona, Jurassic and Triassic granites east of Sierra Nevada. Cretaceous granites in Nevada, California, and Arizona. | High hydraulic gradient. Large vertical movement. Many active faults, Major mining for mineral deposits. High heat flow region. | Some granitic areas in Arizona and Nevada have a relatively low incidence of seismicity, when compared to the California granites. |
| Rocky Mountain Region | Archean granite, Wyoming, and southern Montana. Precambrian granites, Front Range, Colorado. Mesozoic granite, Idaho batholiths, and Montana. | High hydraulic gradient. Large vertical movement. Moderate seismic activity. Large faults bound uplifted blocks. Major mineral deposits. Occurrence of large homogeneous masses of granite. | |
| Lake Superior Region | Precambrian granites, Minnesota, Wisconsin, and Michigan. | Low hydraulic gradient. Little vertical relief. Small number of faults. Very low seismic activity. No volcanic activity. | The most stable region of granite outcrops in the U.S. – part of the N. American Continent stable craton |
| Northern Appalachian and Adirondack Region | Precambrian crystalline rocks, New England and the Adirondacks. | Moderate to low hydraulic gradients. Moderate vertical uplift. No modern fault movement. Low seismic activity. No volcanic activity | |
| Southern Appalachian Region | Precambrian granite, Blue Ridge and Piedmont provinces. | High hydraulic gradient in the Blue Ridge. Intermediate to low hydraulic gradient in the Piedmont. Moderate vertical movement. Very few recent faults. Low seismicity. No volcanic activity. Large masses of granite. | |

* Smedes (1983)
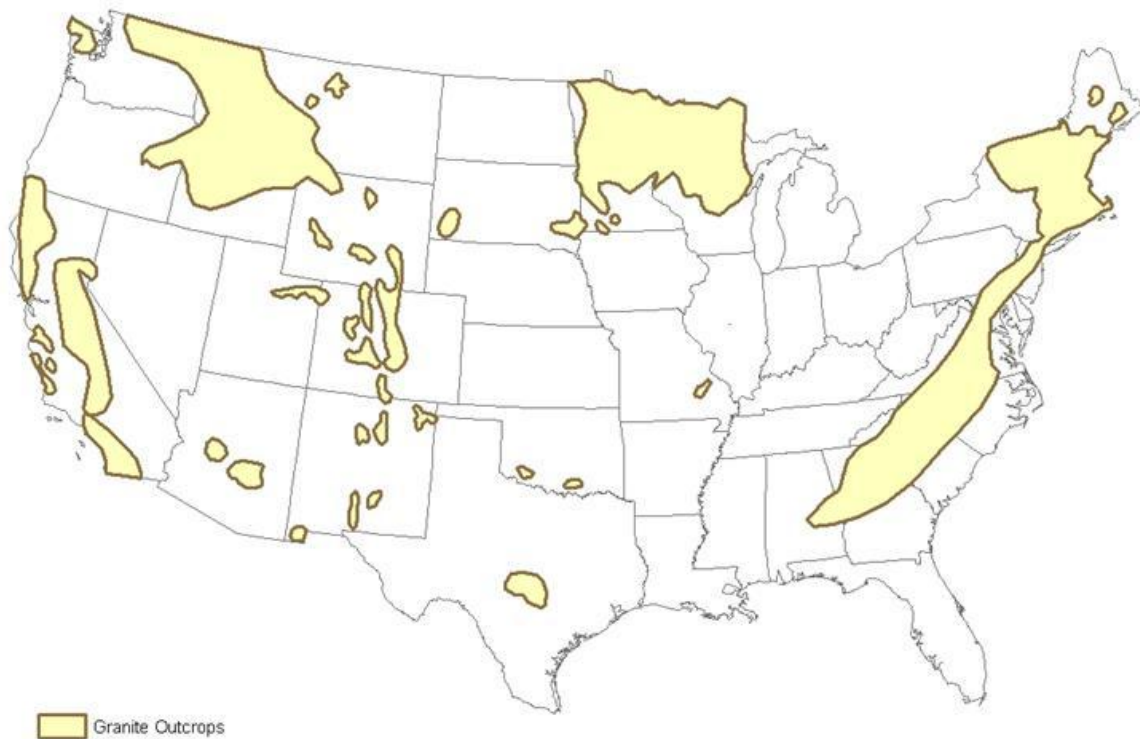
Figure 1-2.  Granite Outcrops in the United States (Bush 1976).

The variation in constituents and emplacement histories of granitic-type rocks can result in significant differences in the rock characteristics. Table 1-3 summarizes some of the most pertinent properties for well-characterized granite formations. These data show that rock strength is high and porosity is generally less than 1%. The wide range of hydraulic conductivity observed is highly correlated with the number, aperture, and connectivity of fractures. Granite bodies are intersected by fracture zones of varying hydraulically conductivity; thus, the higher measurements reflect the effects of fracture zones. The decrease in hydraulic conductivity with depth is consistent with observations of fewer open, interconnected fractures at depth.

Table 1-3. Properties of Well-Characterized Granite Formations.

| Location | Reference | Lithologic Classification | Porosity (%) | Bulk Hydraulic Conductivity (m s$^{-1}$) | Young's Modulus (GPa) | Poisson's Ratio | Compressive Strength (MPa) |
|---|---|---|---|---|---|---|---|
| Barre Granite, Vermont | Krech et al. (1974) | Granodiorite | 0.51 | $< 10^{-10}$ | 46 | 0.23 | 197 |
| St. Cloud Granodiorite, Minnesota | Krech et al. (1974) | Granodiorite | 0.08 | $< 10^{-10}$ | 71 | 0.25 | 282 |
| Westerly Granite, Rhode Island | Krech et al. (1974) | Granodiorite | 0.35 | $< 10^{-10}$ | 50 | 0.21 | 233 |
| Sherman Granite, Wyoming | Touloukian et al. (1981) | Granite | 0.002 | $9.8 \times 10^{-13}$ | | | |
| Granite, Wyoming | Touloukian et al. (1981) | Granite | | | | | 142, 129 |
| Fremont Canyon | Touloukian et al. (1981) | Granite | | | 45 | 0.10 | |
| California | Touloukian et al. (1981) | Granodiorite | | | | | 116 |
| Stone Mountain Granite, Georgia | Touloukian et al. (1981); Hoek & Brown (1980) | Granite | 0.3 | | | | 116 |
| Hardhat Granite | Touloukian et al. (1981) | Granite | | $4 \times 10^{-14}$ | | | |
| Atikokan, Ontario | Gascoyne et al. (1987) | Granite | | $10^{-12} - 10^{-5}$ (<400 m) $10^{-13} - 10^{-10}$ (>400 m) | | | |
| Olkiluoto, Finland | Posiva (2010)[a] | Pegmatitic granite | 0.01 – 0.2 | $10^{-7}$ (near surface) $10^{-11}$ (at depth) | 65 | 0.29 | 108 |
| Laxemar, Sweden | SKB (2006b), Table A-42; SKB (2006d) | Granite to Granodiorite | 0.14 | $10^{-10} - 10^{-5}$ (<200 m) $10^{-11} - 10^{-6}$ (>200 m) | | | 165-210 |
| Forsmark, Sweden | SKB (2006b), Table A-42 | Granite to Granodiorite | 0.09 | | 76 | 0.24 | |

[a] Approximate range of flow porosities, which are 1/10$^{th}$ the approximate diffusion porosities (Tables C-3, C-4, and C-5, Posiva (2010)). Hydraulic conductivity values are from p. 249.

# 2. TECHNICAL BASIS AND CHARACTERIZATION

Disposal of HLW in a repository deep in a granite formation is expected to provide effective long-term ($>10^6$ years) isolation of radionuclides from the biosphere because of mechanical, hydrologic, and chemical conditions favorable to waste form containment. A repository in a granite formation would likely be resistant to inadvertent human intrusion, and the wide distribution of suitable bedrock in the U.S. would increase the chances for a successful repository siting process. More specifically, the attributes of a granite repository include:

- **Minimal rock pressures** – It is expected that excavations in suitable granite formations would be stable and long-lasting (Smedes 1983). Mechanical stability would contribute to worker safety during the pre-closure and operational periods. Waste packages and other engineered barriers would be protected from shear stresses and rockfall during both the operational and post-closure periods. Favorable mechanical characteristics would readily allow waste retrieval should governing regulations so require. There is considerable worldwide experience in the excavation and construction of self-supporting underground openings in crystalline and other hard rocks at the depths considered suitable for a repository. This experience provides confidence that timely, efficient, and cost-effective repository mining can be performed. Specific attention would be needed to avoid major subvertical fracture zones. Moreover, if sited correctly, there would be a low probability of seismicity that could damage the engineered barriers. By protecting the waste packages from rock pressures over long periods of time, the geomechanical properties of granite would strongly contribute to the long term isolation of radionuclides from the biosphere.

- **Low permeability** – It is also expected that a repository excavated in a suitable granite formation would have low rock permeability, which limits groundwater flow. Low interconnected porosity, in combination with much lower kinematic (flow) porosity, limits groundwater flow in granite (Section 2.5.1). Advective flow of groundwater can lead to the enhanced migration of radionuclides away from the repository and towards the biosphere. In granite plutons, lighter freshwater is found near the surface and heavier saline water is found at depth, which is a stable arrangement that tends to reduce vertical groundwater flow. Chemical age dating indicates that groundwater at the depths of granite repository sites is very old, which supports the hypothesis that saline water at depth does not mix readily with surface waters (Section 2.2.2). Hence, any radionuclides migrating from the repository would not necessarily be destined for the biosphere or would reach the biosphere slowly.

- **Favorable chemical environment** – It is expected that a repository excavated in granite formations would have a reducing environment (Section 2.2.2) that would limit corrosion of the waste canisters (Section 2.4.2), contributing to their long term ability to isolate waste. A reducing environment would also limit solubility and/or increase the adsorption of some radionuclides, further delaying the migration of these radionuclides to the biosphere.

- **Low probability of accidental human intrusion** – It is expected that a repository excavated in a granite formation would have a low probability of human intrusion. The

depth of the repository would restrict or prevent access into the closed repository. The site location would likely be chosen such that there are no known significant natural resources or geothermal heat sources nearby that might encourage exploratory drilling. For example, the groundwater at repository depth would likely be of limited use (too salty) or too uneconomical to exploit. Moreover, the repository would be positioned within a region of rock with low permeability, which would be inconsistent with groundwater resource use. For these reasons, it is unlikely that wells would be drilled into the repository in search of potable water or other resources. The depth of the repository would require specialized drilling equipment and, therefore, any drilling would likely be part of a carefully monitored and controlled exploration performed by technologically advanced people (Garisto et al. 2009, Section 5.5). Presumably, such people would be aware of the dangers of radioactive waste and would not heedlessly drill into the repository.

- **Wide availability of potential sites** – Another favorable characteristic of granite is that suitable bedrock for a granite repository can be found in many parts of the country (Section 1.3).

The granite repository concept is developed in this study in view of conceptual designs advanced by Sweden, Finland, and Canada. To establish the boundary conditions for analysis we specify:

- **Waste inventory and description** – The inventory described in Appendix A is placed in the repository for permanent disposal. We recognize the tenets of retrievability inasmuch as they have been incorporated into international repository programs. However, the primary goal of geologic disposal is to permanently isolate waste material from the biosphere. Therefore, although retrievability is facilitated in a granite repository by the long-term stability of granite, for the purpose of this study, retrievability is not a design priority.

- **Geologic setting** – The repository is deep enough below the present land surface to ensure that the waste is not exposed to the biosphere through erosion or shallow groundwater circulation during its period of isolation (e.g., one million years). By siting the repository at least 300 m beneath the present land surface in granite where fractures are sparse and hydraulic conductivity is low, erosion and shallow groundwater circulation will not threaten repository performance.

- **Concept of operations** – Based on disposal concepts in Sweden and Finland, the repository is conceptualized as a network of tunnels in which waste contained in copper canisters are emplaced in boreholes drilled in the tunnel floor. The diameter and length of the emplacement borehole are larger than the waste package to accommodate a clay buffer that surrounds the waste package in the borehole. After emplacement, the emplacement tunnels are backfilled with a mixture of crushed rock and clay.

This study primarily utilizes data and designs advanced by international repository programs. There is a vast amount of descriptive information available. From international experience, emplacement layout and host rock properties are some of the basic parametric selections. Given

a selection of host rock characteristics, waste inventory, and concept of operations, the appropriate FEPs are identified in Section 3 for the generic safety analysis.

As discussed in Section 1.2, we expect that the overall repository waste isolation performance measure of interest would be mean annual dose to a hypothetical individual, with limits set at 0.15 mSv (15 mrem) for $10^4$ years following disposal, and 1 mSv (100 mrem) for the period between $10^4$ years and $10^6$ years. Other details of the regulatory framework, including screening criteria for potentially relevant FEPs, are assumed to be unchanged from those stated in 40 CFR Part 197 and 10 CFR Part 63.

The following subsections examine the major FEPs involved in the performance of a granite repository. They include the U.S. nuclear waste inventory (Section 2.1), geologic setting (Section 2.2), general repository design (Section 2.3), design and function of engineered barriers (Section 2.4), groundwater flow and transport (Section 2.5), and expected evolution of the repository over time (Section 2.6).

## 2.1.  U.S. Nuclear Waste Inventory

The U.S. nuclear waste inventory depends on the future use and potential reprocessing of UNF. With reprocessing, the inventory would consist largely of three types of waste: UNF, existing DOE HLW, and reprocessing HLW. Based on the specific set of assumptions and calculations presented in Appendix A, the total number of waste packages needed to contain these wastes is estimated to be 32,154 for UNF, 5,003 for DOE HLW, and 4,055 for reprocessing HLW. Appendix A includes tables of waste package radionuclide inventories for each waste package type and the half-lives of the radionuclides.

Decay and ingrowth cause the radionuclide inventories to change over time. GoldSim version 10.5 was used to calculate these changes for each waste package type. The resulting calculations are shown in Figure 2-1 (UNF), Figure 2-2 (DOE HLW), and Figure 2-3 (reprocessing HLW).

The thermal power of each waste package is a function of the radioactivity. The thermal power is important to the design of a granite repository because temperatures in the repository should be limited. For example, high temperatures may cause buffer alteration or cementation (Section 2.4.2) (SKB 2011, Section S3.6). To prevent temperatures from exceeding specified limits in granite repository designs, a thermal analyses is needed to determine the necessary waste package spacing (e.g., SKB 2011, Section 5.2.2).
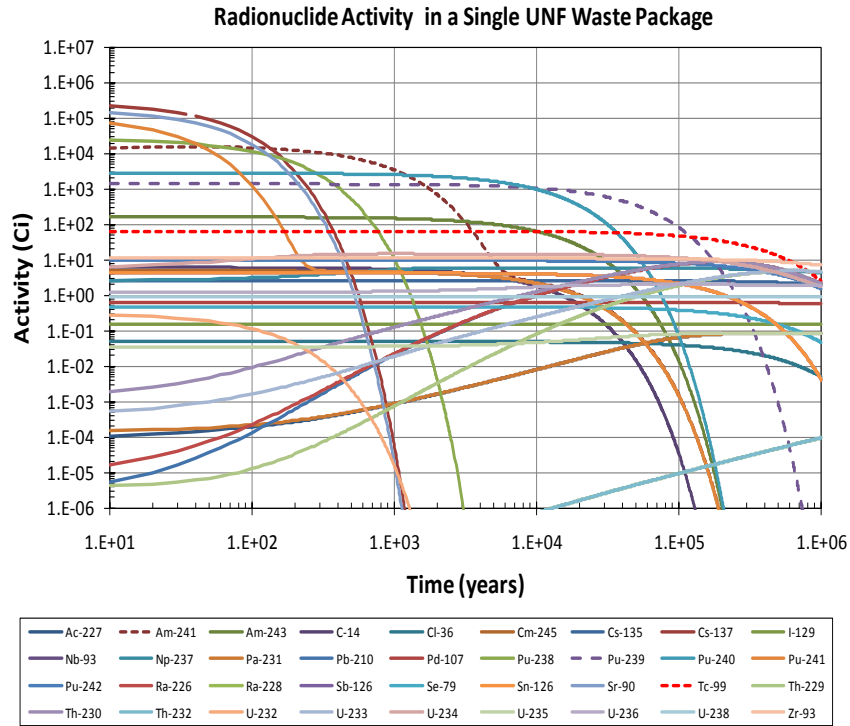
Figure 2-1. Calculated Radionuclide Activity in a Single UNF Waste Package.
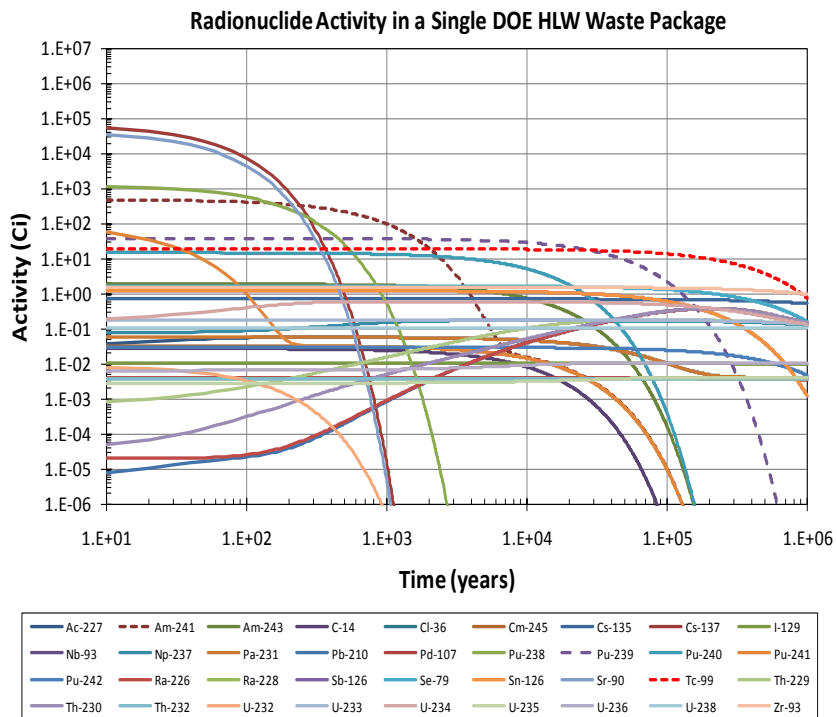


Figure 2-2. Calculated Radionuclide Activity in a Single DOE HLW Waste Package.

**Radionuclide Activity in a Single Reprocessing HLW Waste Package**
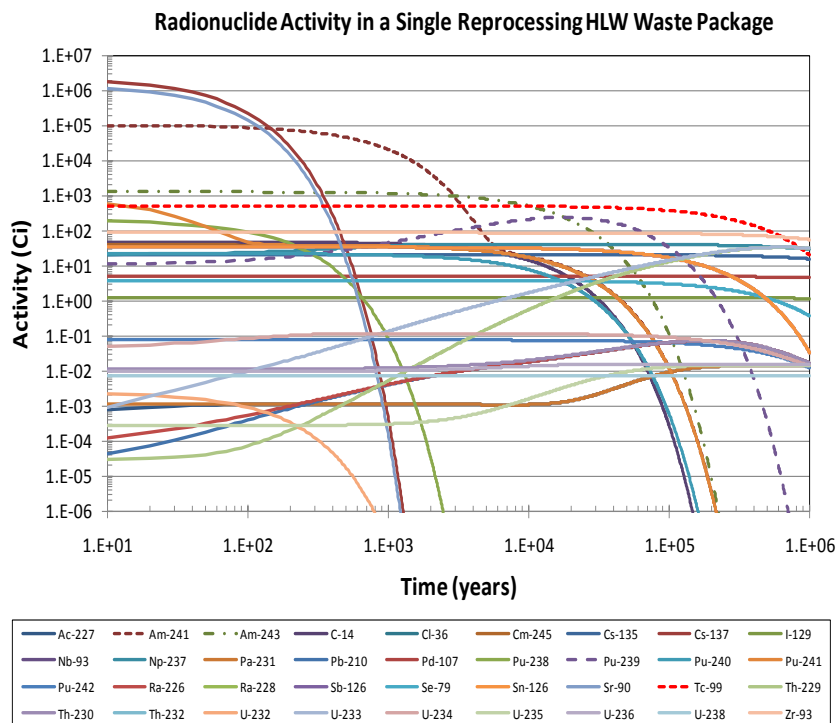


Figure 2-3.  Calculated Radionuclide Activity in a Single Reprocessing HLW Waste Package.

## 2.2.  Geologic Setting

The geologic setting is important to repository siting, design, and performance. Knowledge of the physical and chemical characteristics of the granite and groundwater are needed to select an adequate location, design the layout of the repository, select materials used for engineered barriers, and conduct the post-closure performance assessment.

### 2.2.1.  Siting

The properties of granite formations vary broadly across the U.S. (Section 1.3) and may also vary broadly across granite provinces. Drawing from McKinley et al. (2007), key characteristics to consider in siting a mined geologic repository in granite include:

- **Depth** – The specific isolation zone should be determined based on site specific conditions. Geologic isolation is attained by ensuring significant separation between the repository and the biosphere, which would provide extensive sub-zones for robust seal systems. Rock strength and in situ stress characteristics would also determine a practical and functional mining depth.

- **Thickness** – Maximal thickness of the isolation medium is desired to ensure radionuclide migration does not exceed regulatory criteria or boundaries. Various "minimal" thicknesses have been proposed, generally on the order of 100 m. However, the thickness of the formation is less important than its uniformity and structure.

- **Uniformity and Structure** – The potential repository interval and surrounding rock should be reasonably homogeneous both vertically and horizontally. The related benefits are simpler and more transparent safety and PAs, as well as safer repository mining and operations. Conversely, extensive faulting complicates the characterization, design and analysis, and also increases the uncertainty in safety and PAs.

- **Seismicity** – Repository design and operational issues favor a relatively seismically quiescent region. These areas would also greatly reduce the risks for seismically induced damage to the engineered barrier system (EBS) and the seismic consequences for long term performance.

- **Hydrogeology and Hydrogeochemistry** – Fractures in the vicinity of the repository should be small and sparse, and the bulk hydraulic conductivity of the granite should be low (approximately $10^{-10}$ m s$^{-1}$ or lower). For the KBS-3 disposal concept considered in this report (Section 2.3), the granite should be saturated and the chemical environment should be reducing. Reducing chemical conditions would minimize the corrosion of engineered barriers and waste forms, reduce most radionuclide solubilities, and improve adsorption.

The physical characteristics of granite bodies generally meet these criteria in the following ways (Smedes 1983):

- Crystalline rocks exhibit great strength, mechanical stability, and rock homogeneity, all of which enhance excavation stability. Many mined tunnels/caverns in granitic rocks have remained intact for centuries without additional support.

- Granite is composed largely of silicate minerals, which crystallize at high temperatures. Therefore, it is unlikely that the silicates would be significantly affected by the heat generated from radioactive waste.

- Crystalline rocks are known for their low permeability, with groundwater flow occurring only through fractures. Rock fractures are known to dissipate in number with depth and may be insignificant at repository depths.

- Large quantities of homogeneous granites are found in regions of low seismic activity and are known to have been stable for millions of years.

The chemical environment within granite bodies is also conducive to waste isolation. The effects of reducing conditions on canister corrosion rates, radionuclide adsorption, and radionuclide solubility are addressed in Sections 2.4.2, 2.5.3, and 2.5.5. In addition, the high salinity of groundwater typically observed in deep granitic rocks (next section) would inhibit radiocolloid transport (Section 2.5.2) and could present some resistance to upward flow due to its likely higher density than overlying waters.

## 2.2.2. Chemical Environment

At the depth of a potential repository in granite (approximately 500 m), brackish groundwater typically saturates the fractures and interconnected pores. Table 2-1 gives examples of the compositions of such groundwater at sites in Sweden, Finland, and Canada. Na-Ca-Cl solutions predominate at this depth with total dissolved solids in the range of 1 to 10 g $L^{-1}$ or higher.

Granitic waters at these depths are typically not modern. Isotope data at Forsmark, for example, indicate that water of meteoric origin does not occur below about 200 m (SKB 2006d, p. 109). At Olkiluoto, mixing models indicate that water at a depth of 500 m is approximately 50% formation water, 10% melt water from the Weichselian glaciation (the most recent glacial period from approximately 110,000 to 10,000 years ago), and 40% water from pre-Weichselian Quaternary glacial cycles (Posiva 2010, Figure 6-11). The maximum fraction of Weichselian melt water at Olkiluoto (~50%) is observed at a depth of approximately 150 m. These data indicate that water at repository depths in granite is not well connected to the biosphere.

The redox conditions at the depth of a potential repository in granite are reducing. There are two primary reasons for this. First, as indicated by the analysis at Olkiluoto, there is little mixing of infiltrating waters to depths of 400 to 500 m, except perhaps over periods of hundreds of thousands of years. Second, there is an abundance of oxygen-consuming reactants below the surface. At shallow depths, oxygen is typically consumed by microbial degradation of organic carbon. At Olkiluoto, iron oxyhydroxides are observed in fractures only in the top few meters (Posiva 2010, Section 6.2.5). Below this depth, pyrite and other mineral sulfides are present and react with oxygen, producing sulfate. At approximately 300 m at Olkiluoto a spike in the hydrogen sulfide concentrations indicates reducing conditions are strong enough to reduce sulfate to sulfide. Below 300 m, aqueous concentrations of both sulfate and sulfide drop and the concentration of methane rises. Methane is another strong buffer against the downward transport of oxygen because methanotrophs will use available oxygen to oxidize methane.

Studies in Sweden and Canada also indicate reducing conditions in granite formations. All groundwater samples from Äspö and Stripa were shown to contain dissolved Fe(II) despite prolonged periods of oxygen inflow into the tunnels (SKB 2006d, Section 9.2.5). Microbial respiration was shown at Äspö to consume infiltrating dissolved oxygen in the first 70 m of a major fracture zone even after construction of a tunnel through the fracture zone at that depth caused a 20-fold reduction in the mean residence time (Banwart et al. 1999). In Canada, all Eh measurements from groundwater samples from four research areas in the Canadian Shield indicate redox potentials at or below the Fe(II)/Fe(III) boundary (Gascoyne et al. 1987, Table 3). Reducing conditions are important to repository performance because they prevent rapid oxidation of waste canisters, decrease aqueous solubilities of many redox sensitive radionuclides, and can limit waste form degradation rates.

Table 2-1. Sample Groundwater Compositions in Granite at Depths from 360 to 708 m.

| Parameter | Olkiluoto, Finland | Olkiluoto, Finland | Olkiluoto, Finland | Laxemar, Sweden | Forsmark, Sweden | Pinawa, Canada | East Bull Lake, Canada |
|---|---|---|---|---|---|---|---|
| Borehole | OL-KR20 | OL-KR10 | OL-KR12 | KLX03 | KFM02A | WN-4 | EBL-2 |
| Depth (m) | 360 | 487 | 708 | 380 | 512 | 513 | 538 |
| TDS (g L$^{-1}$) | 10.5 | 22.1 | 49.5 | 2.8 | 9.3 | 7.5 | 2.3 |
| Ionic strength (eq L$^{-1}$) | 0.22 | 0.48 | 1.18 | 0.05 | 0.19 | 0.16 | 0.05 |
| pH | 7.4 | 8.0 | 8.2 | 7.9 | 7.2 | 8.1 | 7.4 |
| Na (mol L$^{-1}$) | 0.11 | 0.21 | 0.36 | 0.03 | 0.09 | 0.07 | 0.03 |
| Ca (mol L$^{-1}$) | 0.03 | 0.09 | 0.25 | 0.01 | 0.02 | 0.03 | 0.01 |
| K (mol L$^{-1}$) | $2.8 \times 10^{-4}$ | $3.6 \times 10^{-4}$ | $4.9 \times 10^{-4}$ | $1.4 \times 10^{-4}$ | $9.0 \times 10^{-4}$ | $5.3 \times 10^{-4}$ | $5.4 \times 10^{-5}$ |
| Mg (mol L$^{-1}$) | $2.6 \times 10^{-3}$ | $1.6 \times 10^{-3}$ | $1.5 \times 10^{-3}$ | $4.4 \times 10^{-4}$ | $9.3 \times 10^{-3}$ | $1.1 \times 10^{-3}$ | $7.0 \times 10^{-5}$ |
| Sr (mol L$^{-1}$) | $1.6 \times 10^{-4}$ | $3.7 \times 10^{-4}$ | $1.1 \times 10^{-3}$ | nr | nr | nr | $3.3 \times 10^{-5}$ |
| Mn (mol L$^{-1}$) | $5.8 \times 10^{-6}$ | $7.3 \times 10^{-6}$ | $9.3 \times 10^{-6}$ | nr | nr | nr | nr |
| Cl (mol L$^{-1}$) | 0.18 | 0.38 | 0.86 | 0.04 | 0.15 | 0.11 | 0.04 |
| SO$_4$ (mol L$^{-1}$) | $2.1 \times 10^{-4}$ | $1.0 \times 10^{-5}$ | $5.0 \times 10^{-5}$ | $1.3 \times 10^{-3}$ | $5.2 \times 10^{-3}$ | $6.6 \times 10^{-3}$ | $1.4 \times 10^{-4}$ |
| CO$_3$ (mol L$^{-1}$) | $5.5 \times 10^{-4}$ | $1.1 \times 10^{-4}$ | $4.0 \times 10^{-5}$ | $3.1 \times 10^{-3}$ | $2.2 \times 10^{-3}$ | $3.5 \times 10^{-3}$ | $5.0 \times 10^{-4}$ |
| SiO$_2$ (mol L$^{-1}$) | $3.6 \times 10^{-4}$ | $2.8 \times 10^{-4}$ | $2.1 \times 10^{-4}$ | nr | nr | nr | $5.4 \times 10^{-5}$ |
| Fe (mol L$^{-1}$) | $2.5 \times 10^{-6}$ | $2.0 \times 10^{-6}$ | $3.8 \times 10^{-7}$ | $8.0 \times 10^{-6}$ | $3.3 \times 10^{-5}$ | nr | nr |
| S(-II) (mol L$^{-1}$) | $5.6 \times 10^{-6}$ | $<3.1 \times 10^{-7}$ | $1.3 \times 10^{-6}$ | $3.0 \times 10^{-7}$ | 0.0E+00 | nr | nr |
| Reference | Posiva (2010), Table 6-6 | Posiva (2010), Table 6-6 | Posiva (2010), Table 6-6 | SKB (2006d), p. 382 | SKB (2006d), p. 382 | Gascoyne et al. (1987), Table 3 | Gascoyne et al. (1987), Table 3 |

nr = not reported

## 2.3. Granite Repository Design

After waste inventory (Section 2.1) and geologic setting (Section 2.2), the third boundary condition governing repository design is concept of operations. Section 2.3 broadly summarizes the concept of operations adopted for this report. A more detailed discussion of EBS features and their functions is provided in Section 2.4.

Because Sweden, Finland, and Canada have advanced concepts for a repository in granite, their experience is used to inform our generic repository design. The conceptual granite repositories for these countries are at depths of 420 m (Finland) and 500 m (Sweden and Canada) (SKB 2006d; Posiva 2010; Garisto et al. 2009).

The reference layout and arrangement of waste packages are also similar among the three countries. Sweden and Finland are studying waste package emplacement in vertical boreholes drilled into the floor in horizontal emplacement tunnels but are also considering emplacement in horizontal boreholes drilled in the tunnel walls (Figure 2-4). Canada is considering these two options in addition to in-room horizontal emplacement (Gierszewski et al. 2004). Each of these concepts includes a clay buffer surrounding waste packages, a mixture of clay and crushed rock to be used as backfill, and various grouting and sealing processes.
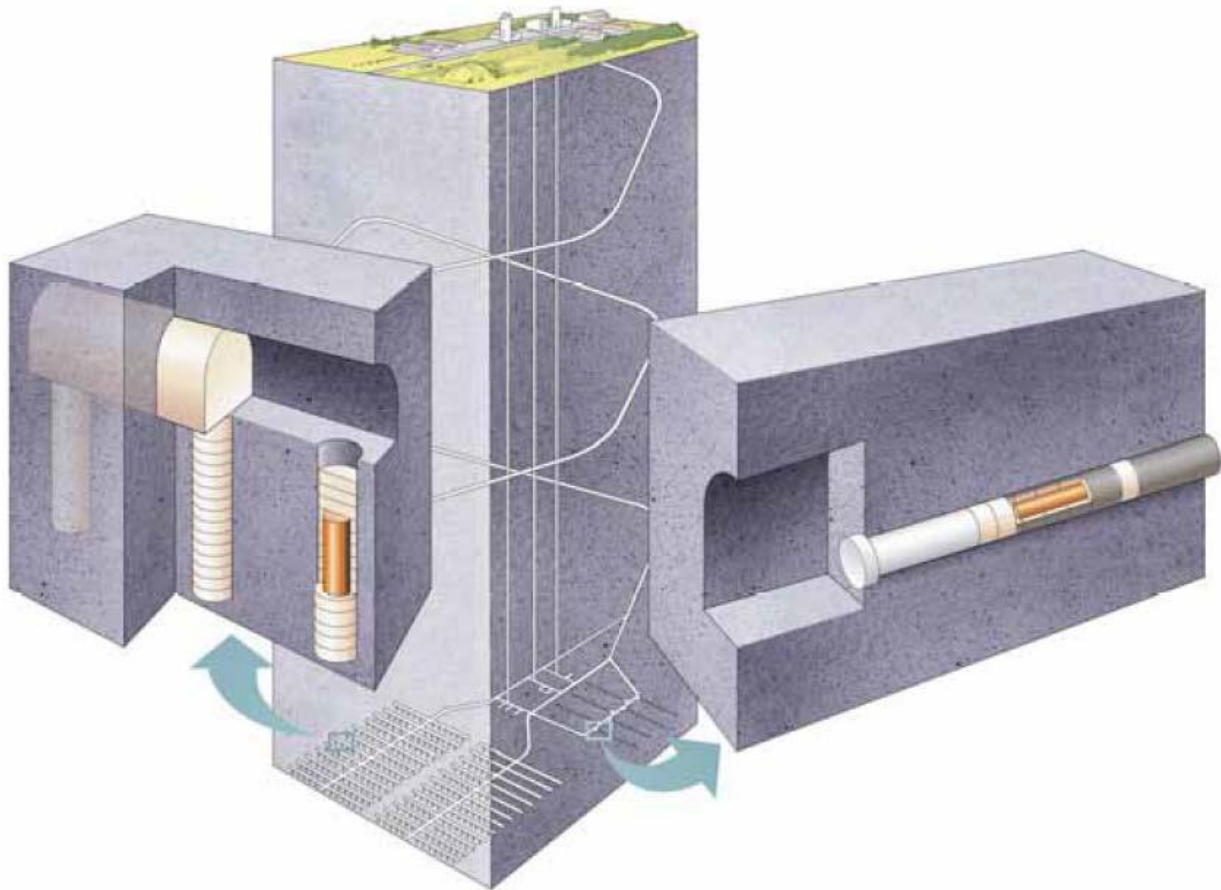


Figure 2-4.  The KBS-3V (left) and KBS-3H (right) Alternatives of the KBS-3 Spent Fuel Disposal Method (from Posiva (2010), Figure 4-1).

A requirement in repository programs is to provide a suitable distance between the emplaced waste and major fractures. Large fractures can provide channels for major water flow and can present rock stability problems. Although major fractures can be avoided in the repository excavation, smaller fractures cannot be. For fractures that intersect the excavations and could contribute significant water inflow, grouting and cementing, anchor bolts, and other supporting measures would be used during repository construction to limit groundwater flow and to ensure operational safety. Waste packages would not be emplaced near any such zones.

During the first few thousand years, the temperature in the repository would be elevated. Waste package thermal loads and spacing within the repositories at the time of emplacement would be

designed to prevent temperatures at the waste package surface from exceeding 100°C. This limit is primarily imposed by the international programs to prevent damage to the buffer (SKB 2011, Section S3.6). To satisfy this requirement in the Forsmark repository design, the emplacement tunnels are spaced about 40 m apart and have vertical disposal boreholes about 6 m apart (SKB 2006d, Table 9-4). For the 6,000 waste packages in the design, this spacing covers an area of approximately one square kilometer. The conceived repository layout splits this area into to several smaller deposition areas connected by transport tunnels (SKB 2011, Figure 5-3).

## 2.4. Engineered Barrier System

Engineered barriers have an important role for waste isolation in fractured, crystalline rock. Whereas the natural setting may attenuate sorptive or redox-sensitive waste species that are released from the repository, the more mobile species (e.g., $^{129}$I) may be transported rapidly by fracture flow. The KBS-3 disposal concept (SKB 2006d), shown in Figure 2-4, is currently accepted worldwide as a reference for disposal in crystalline rock. The Swedish program for used fuel disposal is one of the most advanced internationally, considering investigations in all geologic media, and our current understanding of the likely performance of a U.S. repository in crystalline rock is attributable to the published efforts of Svensk Kärnbränslehantering AB (SKB) and its collaborators from Canada, Finland, France, Spain, Switzerland, and other countries.

The KBS-3 concept calls for disposal of used uranium-oxide fuel in sealed copper canisters (1.05-m diameter) that are emplaced in large-diameter (1.75 m) deposition boreholes drilled into the floor in access tunnels (5.5-m diameter) approximately 500 m below the surface (Figure 2-5). Fuel assemblies are positioned inside a canister insert made of cast iron (Figure 2-6). The space between the canister and borehole walls is filled with a low-permeability buffer material containing bentonite or other swelling clay emplaced initially in its dry, compacted form. In the original KBS-3 concept the boreholes are vertical and extend approximately 7.8 m into the floor of the access tunnel (Figure 2-5). As an alternative, the Finnish and Swedish programs are also considering a design (KBS-3H) in which multiple waste canisters are emplaced with buffer material in horizontal boreholes drilled into tunnel walls; however, further research is needed to fully resolve critical technological issues (SKB 2011, Section 1.1.1; Posiva 2010).

The KBS-3 concept has been further developed and refined since originally proposed, and key aspects have been tested in the laboratory and in situ at scales up to full scale. As currently described (SKB 2006d, Section 4.1.1), the EBS includes the waste form, insert and canister, buffer, bottom plates in deposition holes, access tunnel and its backfill, other openings with backfill, and repository plugs and seals.
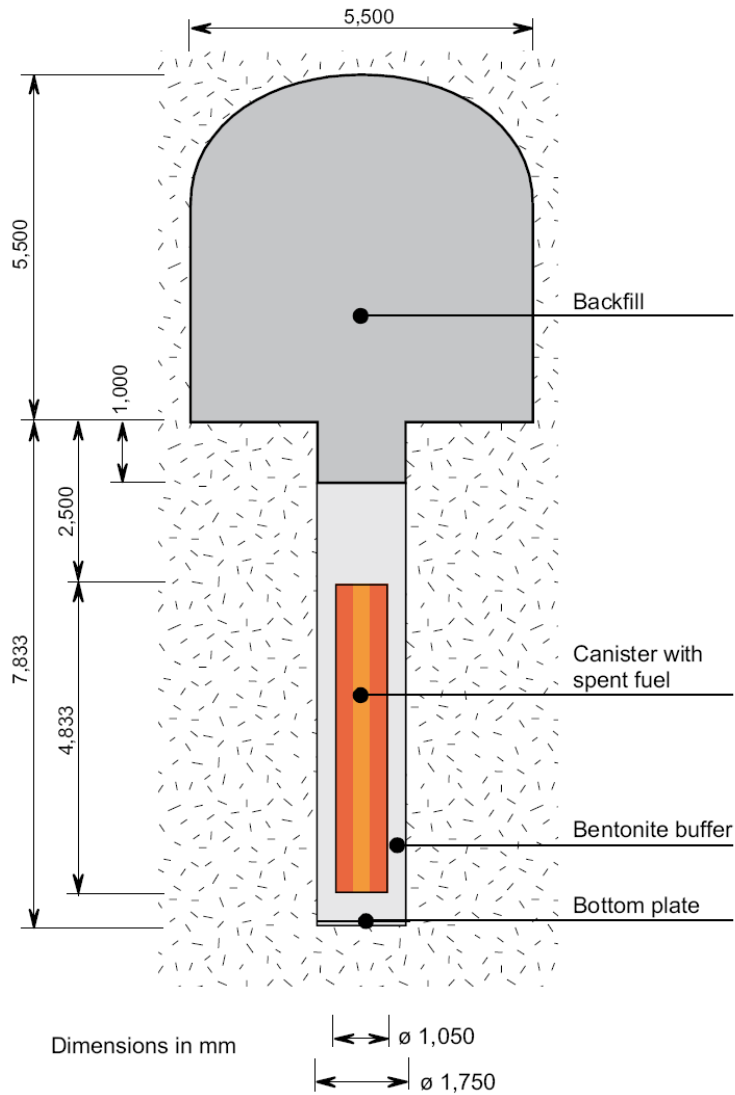
Figure 2-5.  Schematic of KBS-3 Vertical Emplacement Concept (from SKB (2006d), Figure 4-4).

The relative performance of the EBS in a repository in crystalline rock depends on the geologic setting. The KBS-3 concept is being developed for Scandinavian sites where the geologic settings involve sub-sea hydrologic conditions, host rock formations containing chemically reducing mineralization, and a set of future events and processes particular to the setting. Note also that Sweden has deployed interim storage at a centralized used-fuel storage facility, which limits waste heat output at the time of disposal in the repository and is a basic component of the management system.
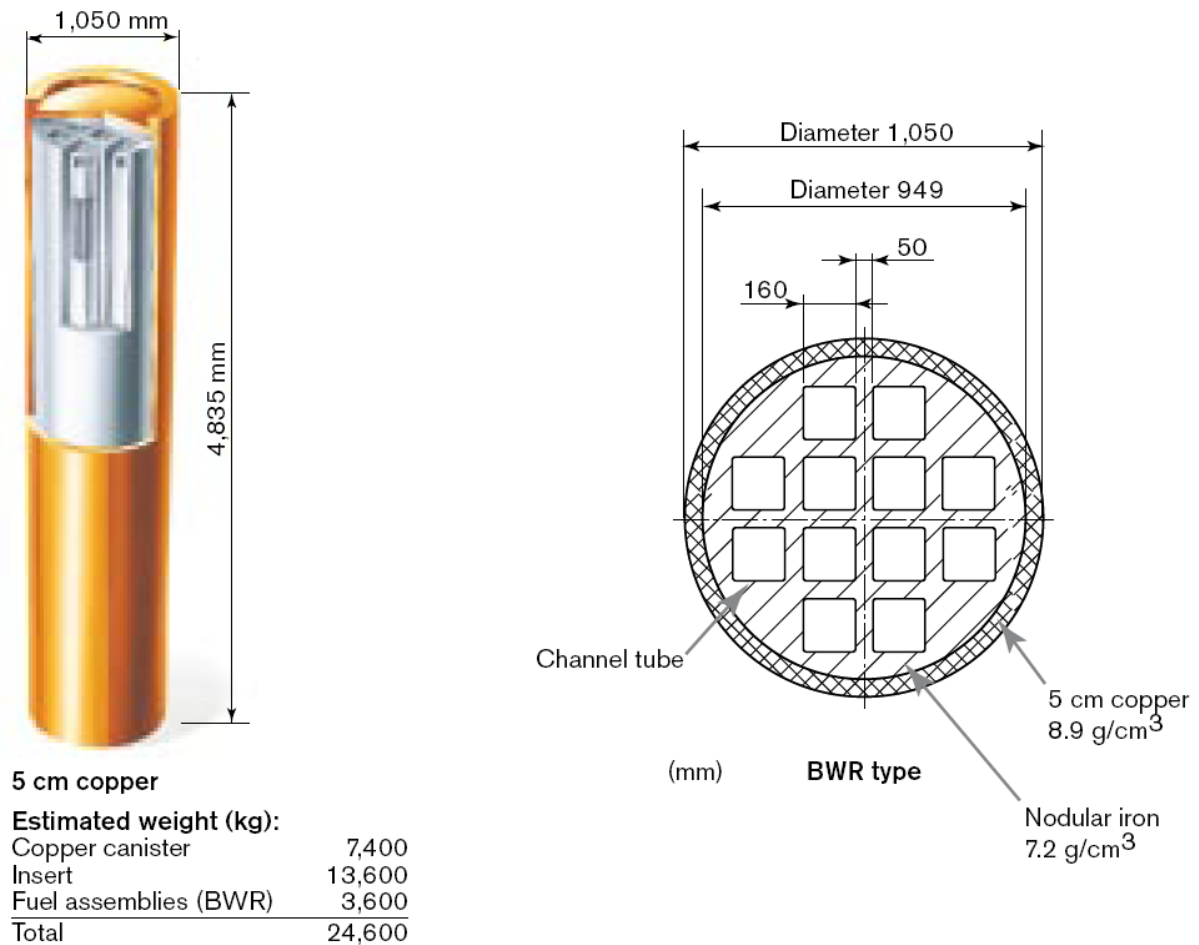
Figure 2-6.  Schematic of Copper Canister and Cast-Iron Insert, as Intended for Boiling Water Reactor (BWR) Used Fuel in the KBS-3 Concept (from SKB (2006d), Figure 4-5).

As discussed below, a general concept of operations has been proposed that uses clay buffer material as a principal element of the EBS and that could be combined with natural barrier performance in a range of geologic settings. Thus, the KBS-3 concept may be appropriate for direct application to a repository in the U.S., or it may be modified to reflect local site conditions and particular future events and processes.

A variant of the KBS-3 concept was proposed by McKinley, Apted, and others as a practical method to achieve optimal waste isolation in fractured, hydraulically saturated, geologic media (McKinley 1997; Apted 1998; McKinley et al. 2000). The prefabricated EBS modular (PEM) concept relies on radial layers of pure bentonite and sand-bentonite buffer materials surrounding a HLW package. The components are assembled and sealed in a thin-walled metal envelope for handling, transport to the repository, and direct emplacement in underground drifts. Swelling of the clay buffer follows failure of the metal envelope, generally from corrosion. Considerable cost savings are possible using a simple design with common materials and prefabrication. Development of the concept continues to the present (McKinley et al. 2006). The concept is applicable to waste disposal in various media in the U.S. including crystalline rock.

The following discussion focuses on the Swedish KBS-3 vertical emplacement concept because of the extensive analysis and literature available.

## 2.4.1. Functional Description of EBS

The following description was provided for the SR-Can preliminary safety assessment (SKB 2006d; SKB 2006a) unless otherwise noted. The EBS components described include the waste form, canister and insert, buffer, bottom plates in deposition holes, access tunnel and its backfill, other openings with backfill, and repository plugs and seals.

- **Waste Form** – Uranium oxide (UOX) or mixed-oxide (MOX) fuels are metal-clad ceramic materials providing mechanical integrity and limited rates of waste form degradation. UOX and MOX fuels retain uranium and other actinides in reduced valence, which limits metal solubility within the EBS while redox potential is low and in reducing groundwater.

- **Canister/Insert** – The canister and insert facilitate handling of used fuel in the repository (Figure 2-6). They provide part of the radiation shielding needed to protect personnel during repository operations, and they dissipate waste heat. Copper and cast iron materials are chemically compatible with the waste form, buffer, and other EBS materials, and they exhibit simple and readily predictable corrosion modes and rates. Both corrode slowly under reducing conditions.

- **Buffer** – The principal buffer function is to limit the transport of aqueous solutes between the waste package and the geosphere. Molecular diffusion is the dominant transport process. For many radionuclides and other solutes, transport through the buffer is further limited by adsorption. The buffer material contains expansive clay as a major constituent and is emplaced in the compacted, dry state. Buffer swelling and transport performance are only slightly or moderately sensitive to temperature and saline solutions (i.e., seawater). Buffer performance is insensitive to the presence of microbes or to microbial activity. Once hydrated the buffer not only limits outward transport but also limits the rates of inward transport of dissolved reactants that control degradation of the canister and its contents (Section 2.4.2).

- **Bottom Plates** – The bottom of each deposition hole is leveled using cast-in-place concrete (low pH cement). The plates provide structural support and centering for installation of the buffer and canisters and a sump for collecting and removing water. The plates can be isolated from much of the buffer by a thin sheet of inert material such as copper.

- **Access Tunnel Backfill** – Backfill in the access tunnels serves two major functions: mechanical constraint of the swelling buffer in deposition boreholes and limited aqueous transport of radionuclides. Mechanical constraint is provided by controlling backfill composition and compaction at installation. Limited aqueous transport is provided by controlling backfill transmissivity so that it is similar to the host rock. Also, the backfill is intended to swell to fill any voids present at its installation and to apply pressure to the disturbed rock zone around the tunnel opening to limit permeability to fracture flow.

- **Other Openings, Backfill, and Seals** – Backfill and seals in other parts of the repository limit aqueous transport of radionuclides by controlling both inflow and outflow. Seals and plugs control hydraulics at key interfaces or intersections in the facility while limiting permeability of the disturbed rock zone and providing structural support to the access tunnel backfill. The design approach for shaft and drift seals would be similar to that used for the WIPP (Hansen and Knowles 2000).

## 2.4.2. EBS Performance

This discussion focuses on the processes that both support and degrade the EBS functions described above. Much of the information for clay buffer processes is from the SR-Can assessment (SKB 2006d; SKB 2006a) unless noted otherwise. This description supports the discussion of FEPs in Section 3.

**Buffer Swelling and Mass Distribution** – The processes that control buffer swelling and mass distribution are fundamental to EBS performance. During initial hydration there is a strong water potential gradient between liquid groundwater and the buffer in its initial state. Before hydration the buffer is unsaturated and has high bulk permeability. The buffer hydrates non-uniformly along joints between the blocks of compacted clay and thus is likely to retain trapped air. Over time the buffer achieves a more uniform state of hydration, in part because swelling pressure displaces clay and water into less hydrated parts. Trapped air eventually dissolves in the buffer pore fluid at the hydrostatic groundwater pressure associated with repository depth.

Buffer material could exhibit, or has been described as exhibiting, stages of deformation as it hydrates and adjusts to in situ mechanical and hydrologic conditions: primary (pore pressure controlled), secondary (volumetric creep of the clay framework), and tertiary deformation in which the clay softens in response to deviatoric stress. The buffer can deform to adjust to small movements of the waste package or of the host rock.

Mechanical pressure effects during buffer hydration and swelling may contribute to formation of voids (piping) and may also displace waste canisters in addition to loading them by external pressure. Upward displacement of clay buffer material may occur within the annulus around the canister and into the overlying access tunnel backfill. Buffer material may also extrude into fractures, possibly contributing to erosion if the extruded material is exposed to flowing groundwater. Pressure in the buffer may displace rock wedges bounded by the borehole wall and natural fractures into the overlying tunnel backfill. Thermal excursions of buffer material pore pressure are potentially significant when the buffer is fully saturated if drainage from the buffer is impeded (which is unlikely in fractured rock with direct communication to the overlying access tunnel backfill). Each of these effects from swelling has been evaluated by simulation or empirical testing and is accommodated in the design, typically by ensuring that the buffer has sufficient saturated density. Swelling pressure has no significant effect on the waste canisters.

Apparent buoyancy of canisters may be caused by higher buffer density and swelling pressure below the canister than above (even though the canister has greater bulk density than the hydrated buffer). Because volumetric creep is likely to be maximal under the canisters and its potential rate is considered to be no greater than deviatoric creep observed in laboratory tests, the

corresponding displacement of the canister throughout the period of repository performance is anticipated to be insignificant to waste isolation.

Eventually the canisters will fail due to corrosion after hundreds of thousands or millions of years (SKB 2006d, Section 9.4.9), and buffer material could extrude directly into the failed canisters. The impact on buffer performance is considered insignificant due to limited internal canister void volume and because of the increased volume occupied by corrosion products from the cast iron insert. Corrosion products from degradation of the canister, insert, or waste will expand against the buffer, increasing the effective swelling pressure.

The foregoing discussion is equally applicable when the buffer is hydrated or otherwise exposed to fresh groundwater, seawater, or high-Ca water (such as could be leached from cement). High salinity or high-Ca pore water can reduce swelling pressure and increase hydraulic conductivity, but this is significant to buffer performance only for saturated density less than approximately 1,800 kg m$^{-3}$.

The current understanding of clay buffer phenomenology is based on extensive laboratory and in situ testing (Stripa Buffer Mass Test, Aspö tests, FEBEX test in Switzerland). Long-term, full-scale tests are presently underway to challenge and confirm the current understanding of clay buffer evolution.

**Piping and Erosion** – The possibility that voids in the buffer may form and connect (i.e., piping) during initial hydration is controlled by competition among the rates for contributing processes: buffer swelling, development of groundwater pressure, and erosion of the buffer material by flow. Analysis has shown that for the likely range of water inflow rates, the rate of swelling is sufficient to prevent piping.

Piping eventually terminates from swelling pressure, but erosion may be continuous. Erosion of smectite at the buffer-groundwater interface may be possible if the concentration of divalent cations is less than 1 mM (See discussion of glaciations, Section 2.4.3.). The performance impact of erosion has been evaluated using the metric of buffer mass removed, and up to 100 kg may be lost per canister without degrading buffer performance. Even if more were lost, advective transport conditions would not necessarily prevail in the buffer (SKB 2006d, Section 9.3.9). Piping and erosion are currently under investigation in the Prototype Repository long-term test (Sweden).

**Gas Generation and Escape** – Corrosion of the canister and insert could potentially produce enough hydrogen and other gases to create a reservoir of gas at sufficient pressure to penetrate the buffer episodically. The process for associated aqueous release of radionuclides involves failure of the bottom of the canister, gas pressure build-up within the canister due to corrosion of the cast-iron insert, and expulsion of collected liquid from the bottom when the gas pressure exceeds the buffer swelling pressure. Such pressure release is phenomenologically complex and is understood mainly from small-scale lab investigations. Analysis shows that such events would be episodic with little water released, and re-sealing of the buffer would occur between events. Increased gas pressure is most likely to occur from corrosion of the cast iron insert and would be expressed on both the inside and outside of affected canisters. Yet to be evaluated is the use of

more corrosion resistant materials (e.g., stainless steels) for the canister or its internals to slow the rate of gas production to that which could diffuse through the buffer.

**Canister and Insert Corrosion** – The copper canister can corrode from exposure to oxygen initially present in the buffer or in groundwater, from radiolytic acid species, and from hydrogen sulfide formed from the dissolution of pyrite in the buffer or host rock. Potential corrosion damage from these effects has been evaluated, mainly from mass balance and buffer transport considerations, and found to be minor (SKB 2006d, Section 10.6). Canister corrosion could proceed most rapidly in the unlikely condition that buffer failure exposes the canister directly to oxygenated or sulfide-containing groundwater. Even under such conditions, corrosion would be slow and canister penetration would not be significant for at least 100,000 years (SKB 2006d, Section 9.4.9). The insert would corrode once the canister wall is penetrated and would limit the availability of oxidizing species that contribute to radionuclide solubility and mobility

**Radionuclide and Colloid Transport** – Advective aqueous transport of radionuclides in the clay buffer is negligible for hydraulic conductivity less than $10^{-12}$ m s$^{-1}$. For this condition transport is dominated by diffusion with a margin on the order of two orders of magnitude. Diffusive transport is readily represented using known ranges for species-specific diffusion coefficients (Section 2.5.4). Ion exchange for cations (interlayer sites) and surface complexation (edge sites, oxyanions, etc.) are the predominant mechanisms for adsorption. Radionuclide adsorption in the clay buffer can be represented using system-relevant, species-specific distribution coefficients (Section 2.5.3).

Buffer swelling reduces pore size and homogenizes the pore size distribution (for volumetrically constrained swelling). This reduces colloid mobility to the point where a saturated density criterion (1,650 kg m$^{-3}$; SKB (2006a, Section 1.7)) substantially eliminates potential colloid transport.

The composition of water that initially permeates the buffer affects the composition of buffer pore water and the chemical conditions for transport. Thus, speciation and solubility for radionuclides in the buffer are affected by the initial clay composition and the composition of groundwater that causes initial hydration.

Surface diffusion (enhanced relative to calculated apparent bulk diffusion using system-relevant adsorption data) is observed for some cations. The concept is that increased cation concentrations in the surface layer allow increased concentration gradients. Also, if adsorption decreases due to reduced availability of adsorption sites, a similar result to enhanced diffusion could be observed. Surface diffusion is converse behavior to anion exclusion whereby anion concentrations are decreased in the double layer so that anion concentration gradients are also decreased. Anion exclusion is potentially important to waste isolation performance for non-sorbed anionic species (iodide, selenium oxyanions, etc.). Anion exclusion has been associated with dry bentonite density in the range 1,200 to 1,500 kg m$^{-3}$ or greater. At higher ionic strength it is less important because the controlling dimension of the electrical double layer is decreased.

**Alteration of Buffer Clay** – Illitization may be important if there is sufficient K$^+$ introduced during hydration along with environmental conditions (e.g., temperature) favoring reaction. In

general, illitization has been found to be an insignificant prospect because of the limited availability of $K^+$ and the limited temperature (<150°C) and duration of the thermal period.

**Alteration of Buffer Accessory Phases** – Other materials or minerals present in the clay buffer in potentially significant amounts include carbonate, pyrite, and organic matter. Carbonate controls the pH, while pyrite and organic matter, if present, control redox potential. Buffered (near-neutral) pH and reducing conditions favor slow canister corrosion and decreased radionuclide mobility.

The presence of silica and its dissolution and precipitation may be important to buffer performance because cementation can result. Cementation decreases swelling pressure and increases rigidity, which may promote formation of voids or fractures. Source minerals for silica include feldspar and quartz. The existence of natural clays for millions of years in situ without cementation or other degradation indicates that buffer cementation will not be important at least for ambient temperature conditions. During the thermal period, prograde silica solubility and faster dissolution of source minerals may increase cementation in colder regions. However, the limited duration of elevated temperature and the availability of water for reaction and transport probably limit the importance of thermally driven silica cementation. Cementation in the buffer is an important uncertainty associated with EBS performance.

**Backfill Characteristics** – Backfill may be composed of a mixture of crushed rock and clay material with as little as 30% clay. The thermal, hydrologic, and rheological properties of such a mixture differ from the clay buffer. Swelling response is less than a pure clay because the achievable density of the clay fraction is less and the backfill contains larger particles and thus larger pores. The sensitivity of clay properties (and thus bulk properties) to the effects of salinity, inhomogeneity, etc., is greater. Similar observations are likely if the backfill is selected to be an unmixed, clay-rich geologic material. The SR-Can assessment assumes that backfill will have hydraulic conductivity no greater than that of the host rock, or $<10^{-10}$ m s$^{-1}$. Because this value is larger than the threshold of $10^{-12}$ m s$^{-1}$ for diffusion-dominated transport in the buffer, radionuclide transport in the backfill may occur by advection as well as diffusion.

Piping and erosion of the backfill during initial saturation affect flow properties locally because of the amount of water involved to resaturate the entire drift from a few points of water entry. However, these channels are expected to heal when full saturation is reached. Depending on the backfill material selected, the maximum swelling pressure may be less than the hydraulic pressure driving water inflow, especially during early stages of saturation, and piping may result. Backfill homogeneity is an objective in drift construction and sealing but difficult to control in practice due to variations in settlement and compaction that can lead to flow channeling.

### 2.4.3. Events That Impact EBS Performance

Application of the KBS-3 disposal concept in northern latitudes where Pleistocene glaciation occurred, presents the likelihood that a glacial climate will return during the repository performance period. Return of continental glaciers is part of the expected future in the SR-Can assessment (SKB 2006d, Section 9.4). The associated and potential impacts on the EBS include

- decreased temperature and the possibility of freezing;

- larger hydraulic gradient, increased flow velocity, and increased hydrostatic pressure on the canisters due to surface relief;

- deep penetration of melt water;

- erosion of the clay gel by low-salinity groundwater (fresher than seawater, and in particular, less concentrated in divalent cations) and transport of clay away from the buffer; and

- faulting and fracturing caused by the increase in groundwater pore pressure and the isostatic response of the lithosphere to glacial loading.

Freezing of pore water in clay will not occur at temperatures above approximately -5°C because of the reduced activity of water in the electrical double layers adjacent to clay crystal sheets (SKB 2006d, Sections 9.4.3 through 9.4.8). The possibility that this temperature limit would be reached is insignificant because of the repository depth.

The hydraulic effects from a kilometer or more of glacial surface relief would affect flow conditions and the chemistry of groundwater; however, the residual effects on the hydraulic structure and mineralization of the aquifers would be limited to present-day conditions (which have resulted from more than two million years of Pleistocene exposure). The effects would be at maximum during glacial advance or retreat when the ice margin is in the repository vicinity. The potential for buffer failure due to erosion by low-salinity groundwater and mobilization of radionuclides in a relatively oxidizing environment are evaluated in detail in the SR-Can assessment (SKB 2006d, Sections 9.4.6 through 9.4.8).

Faulting will likely occur along existing discontinuities, especially those which responded to previous glacial loading and unloading. These discontinuities would be mapped and avoided during repository development. Some new faulting would be expected in response to glaciation; however, the displacements likely to occur on new faults would likely be much smaller than on existing faults. A similar finding was reached for displacements on "unknown" faults in the Yucca Mountain repository block (Gross 2007, Section 6.11). Given the buffer thickness (0.35 m on the radius; Figure 2-5) displacements of less than 10 cm are not expected to cause canister failure nor significantly disrupt buffer function. The likelihood of new faults with greater than 10 cm displacement, intersecting vertically emplaced waste packages, would likely be insignificant (SKB 2006d, Sections 9.4.5). Probabilistic treatment of existing fracture zones in the Forsmark region yields estimates in the range of 0.001 to 0.04 for the mean number of canisters damaged during the initial 120,000 year glacial cycle in a worst case seismic event in which all 6,000 waste canisters are unsuitably positioned.

## 2.5. Flow and Transport

The flow of groundwater affects the performance of engineered barriers and the transport of released radionuclides. Should a waste package fail, radionuclides would be released to the groundwater, the primary medium for radionuclide transport from the repository to the biosphere. This section focuses on groundwater flow and the processes related to radionuclide transport.

## 2.5.1. Groundwater Flow

The major parameters affecting groundwater flow are captured in Darcy's Law,

$$q_x = -K \frac{dh}{dx}$$

(Eq. 2-1)

where $q_x$ is the specific flux (m s$^{-1}$), $K$ is the hydraulic conductivity (m s$^{-1}$), and $dh/dx$ is the hydraulic gradient. Specific flux is the volumetric flux (m$^3$ s$^{-1}$) divided by the bulk cross-sectional area (m$^2$). Hydraulic conductivity is a lumped parameter related to the permeability of the porous medium, fluid density, and fluid dynamic viscosity (Freeze and Cherry 1979, p. 27).

For a saturated clay buffer, $K$ is less than $10^{-12}$ m s$^{-1}$ (Section 2.4.2). The low hydraulic conductivity is primarily due to the small pore size distribution and high tortuosity of the flow path. The porosity of the clay buffer (0.4 to 0.5) implies that the mean pore velocity is limited to approximately twice the value of the specific flux, which is a key reason that diffusion controls solute transport through the clay.

For granite at depths of at least 400 m, $K$ is commonly in the range of $10^{-10}$ to $10^{-13}$ m s$^{-1}$ (Table 1-3). Sparsely fractured granite at this depth can have flow (kinematic) porosities on the order of $10^{-4}$ and diffusion porosities on the order of $10^{-3}$ (Posiva 2008a; SKB 2006b). The flow porosity is the volume through which water flows relative to the bulk volume of rock. For granite, essentially all flow occurs through interconnected fractures. The much larger diffusion porosity in granite consists of both matrix and dead-end fracture porosity connected to the flow porosity. The extremely low flow porosity of deep granite causes the velocity of water in the interconnected fractures to be orders of magnitude higher than the specific flux. Consequently, advection may dominate solute transport in the flow porosity while diffusion dominates transport in the diffusion porosity (Section 2.5.2).

Based on the low hydraulic conductivity of granite host rock and the likely low hydraulic gradient in the vicinity of a well-sited repository, groundwater flow at repository depth would be quite limited. Transmissive fractures at this depth may have an average spacing of more than 100 m as found at Forsmark (SKB 2011, Section S2.2). As a result, complete resaturation of the backfill and buffer after repository closure may not occur for hundreds or thousands of years (SKB 2011, Section S4.1). Although capillary and osmotic processes would help to saturate the clay buffer, the low hydraulic conductivity of the buffer would inhibit the flow of water contacting the waste packages.

## 2.5.2. Radionuclide Transport

One-dimensional radionuclide transport through porous media with sorption and decay is described by the following equation (de Marsily 1986, Equation 10.3.3; Schwartz and Zhang 2003, Equation 23.27)

$$R_f \frac{\partial C}{\partial t} = D_x \frac{\partial^2 C}{\partial x^2} - v_x \frac{\partial C}{\partial x} - R_f \lambda C$$

(Eq. 2-2)

where

$C$   = aqueous radionuclide concentration (mg L$^{-1}$)
$v_x$   = average groundwater velocity (m yr$^{-1}$)
$D_x$   = coefficient of hydrodynamic dispersion (m$^2$ yr$^{-1}$)
$R_f$   = retardation factor
$\lambda$   = radioactive decay constant (yr$^{-1}$).

The retardation factor incorporates adsorption effects, and the coefficient of hydrodynamic dispersion incorporates mixing and diffusion. These coefficients are addressed in Sections 2.5.3 and 2.5.4.

Eq. 2-2 describes solute transport through a porous medium represented as a single continuum. To model transport in a dual porosity medium, such as fractured granite, a dual porosity approach is typically taken. Transport through interconnected fractures is generally dominated by advection while transport into the connected diffusion porosity is dominated by diffusion. In each domain, the same equation (Eq. 2-2) applies, but the values of the parameters may differ widely. In particular, the value of $v_x$ within the diffusion porosity is zero, and the value of $D_x$ is reduced to the molecular diffusion component (Section 2.5.4).

In fractured granite, the diffusion porosity acts to retard solutes relative to solute velocities within the flow porosity. This diffusion-related retardation effect applies to the transport of both sorbing and non-sorbing solutes. Thus, for a fractured granite with a flow porosity of $10^{-4}$ and a diffusion porosity of $10^{-3}$ (as assumed for the sparsely fractured granite in the safety analysis presented in Section 4), solutes would spend an average of ten times as much time within the stagnant domain of the diffusion porosity than within the flow porosity (assuming the solutes are non-sorbing or equally retarded by adsorption within each domain).

Radionuclides associated with colloids add further complexity to the set of transport equations. The transport of colloids differs from the transport of aqueous species because it is affected by additional processes associated with pore-size exclusion, filtration, straining, and colloid stability. The effects of these processes can facilitate or retard overall radionuclide transport. For this report, colloidal transport is not simulated. The saturation density of the clay buffer is expected to prevent colloid transport through the buffer (Section 2.4.2), and the high salinity of the groundwater is expected to significantly limit colloid stability in the granite host rock (Buck and Sassani 2007, Section 6.3.9). However, because dilute groundwater could potentially penetrate to repository depths (for example, under the extreme conditions potentially imposed by the retreat of an ice sheet), and because the stability, behavior, and generation of potential colloids are highly dependent on local conditions, colloidal transport would need to be considered thoroughly in a site-specific performance assessment.

### 2.5.3. Radionuclide Adsorption

The retardation factor $R_f$ is the ratio of the average velocity of water relative to the average velocity of the solute. If the extent of sorption is independent of solute concentration (i.e., linear sorption), the retardation factor may be described by

$$R_f = 1 + \frac{\rho_d k_d}{\phi_w} \qquad \text{(Eq. 2-3)}$$

where $k_d$ is a conditional distribution coefficient describing the extent of adsorption and $\rho_d$ and $\phi_w$ are the dry bulk density and connected volumetric water content of the porous medium, respectively. The distribution coefficient is conditional because its value is affected by the type of immobile solid sorbent, sorbent specific surface area, temperature, pH, redox conditions, ionic strength, and concentrations of other solutes.

Adsorption of radionuclides to corrosion products, buffer materials, granite surfaces, and secondary fracture minerals has the potential to significantly retard the transport of radionuclides relative to water. For many radionuclides, adsorption would be enhanced by the prevailing reducing conditions.

Table 2-2 lists a set of distribution coefficients for radioelement adsorption onto bentonite under reducing conditions in the presence of groundwater similar to the compositions in Table 2-1. In addition to redox conditions, the adsorption of many of these radionuclides onto bentonite or montmorillonite depends heavily on ionic strength and/or pH. The $k_d$ values in this table indicate that many radionuclides would be strongly retarded by the buffer. For bentonite having a dry bulk density of 1,600 kg m$^{-3}$ and porosity of 0.44, a solute with a $k_d$ value of 1.0 m$^3$ kg$^{-1}$, such as Pu, would have a retardation factor of 3,600. Although transport through the bentonite would be dominated by diffusion rather than advection, this calculation implies that approximately 1/3,600$^{th}$ of the Pu in the bentonite would be aqueous and able to diffuse at any given moment. Based on the $k_d$ values in Table 2-2, adsorption to the bentonite buffer would significantly delay the release of many radionuclides from the EBS.

Adsorption in near-field and far-field granite occurs on fracture walls, fracture minerals, and in the granite matrix. Table 2-3 lists distribution coefficients adopted by the Finnish repository program for modeling adsorption in the far-field granite at Olkiluoto. These values are based on laboratory measurements and consider the site-specific rock types, fracture minerals, and estimated dilute/brackish groundwater composition in the geosphere. Values for Sb and Pb shown in Table 2-3 were not included in the Finnish study but were obtained from the granite adsorption experiments of Ikeda and Amaya (1998) taking into account similar chemical conditions.

Table 2-2.  Selected Bentonite $k_d$ Values for the Chemical Conditions of a Generic Granite Repository.

| Element | $k_d$ (m$^3$ kg$^{-1}$) | Source/Notes |
|---------|------------------------|--------------|
| Ac [a] | 10 | Baston et al. (1999), see Am |
| Am | 10 | Ikeda and Amaya (1998) (high µ,[b] pH 5-10, Eh -220 mV) |
| C, Cl | 0 | Adsorption low, assumed non-sorbing |
| Cm | 10 | Baston et al. (1999) |
| Cs | 0.1 | Mucciardi et al. (1979) (montmorillonite, high µ, high Ca, pH 7-9.3) |
| I | 0 | Mucciardi et al. (1979) (montmorillonite, high µ, high Ca, pH 7.4-8.4) |
| Nb | 3 | Ikeda and Amaya (1998); Erdal (1977); Taki and Hata (1991) |
| Np, Pa [a] | 0.1 | Kitamura et al. (2002); Ashida et al. (1999) (pH 8-9, Eh -550 to -400 mV, µ =1M) |
| Pb | 10 | Ulrich and Degueldre (1993); Ikeda and Amaya (1998) (high µ, pH 5-8.5) |
| Pd | 3 | Tachi et al. (1999b) |
| Pu | 1 | Mucciardi et al. (1979); Ames et al. (1981) (high µ, pH 7-9) |
| Ra | 1 | Tachi and Shibutani (1999) for solution/solid ratio > 100; Ames et al. (1983) |
| Sb | 0.1 | Ikeda and Amaya (1998) (low Eh, high µ, bentonite) |
| Se | 0.03 | Tachi et al. (1999a) |
| Sn | 30 | Oda et al. (1999) (depends on pH) |
| Sr | 0.01 | Mucciardi et al. (1979) (bentonite, high µ, high Ca) |
| Tc | 10 | Baston et al. (1999) (high µ, high Na, high Ca, Eh ~ -400 mV, pH 8-10) |
| Th | 3 | Baston et al. (1991); Ueta (1998) (high µ) |
| U | 10 | Baston et al. (1999) (high µ, high Na, high Ca, Eh ~ -400 mV, pH 8-10) |
| Zr | 30 | Rancon and Rochon (1979) (depends on pH) |

[a] $k_d$ values for Ac are set equal to those of chemically similar Am. $k_d$ values for Pa are set equal to those of chemically similar Np.

[b] ionic strength

Table 2-3.  Granite Matrix $k_d$ Values Used in Posiva (2010, Table 6-9) for Dilute/Brackish Groundwater.

| Element | $k_d$ (m³ kg⁻¹) |
|---|---|
| C, Cl, I | 0 |
| Se | 0.0005 |
| Pd, Sn | 0.001 |
| Sr | 0.005 |
| Nb | 0.02 |
| Am, Cm, Ac [a] | 0.04 |
| Pa, Tc, Cs | 0.05 |
| Sb | 0.1 [b] |
| U | 0.1 |
| Np, Th, Ra, Zr | 0.2 |
| Pu | 0.5 |
| Pb | 1 [b] |

[a] $k_d$ values for Ac are set equal to those of chemically similar Am.

[b] Ikeda and Amaya (1998)

## 2.5.4.  Radionuclide Dispersion

The coefficient of hydrodynamic dispersion captures the effects of mechanical mixing and molecular diffusion and is calculated from the equation

$$D_x = \alpha_x v_x + D_b \qquad \text{(Eq. 2-4)}$$

where $\alpha_x$ is the longitudinal dispersivity (m) and $D_b$ is the pore diffusivity (m² yr⁻¹), which is also known as the bulk diffusion coefficient. The first term in Eq. 2-4 is the mechanical component and the second term is the molecular diffusion component. $D_b$ is a function of the solute diffusivity in aqueous solution $D_w$ (m² yr⁻¹) and the geometry of the pores.  The relationship used in this study is

$$D_b = \frac{D_w \delta}{\tau^2} \qquad \text{(Eq. 2-5)}$$

where $\delta$ is constrictivity and $\tau$ is tortuosity (Ochs et al. 1998).[1] Like $D_x$, $D_b$ does not include the effects of adsorption. For experimental measurements of diffusivity in porous media, $D_b$ is determined from the relationship

$$D_b = D_a R_f \qquad\qquad \text{(Eq. 2-6)}$$

where $D_a$ is the measured apparent diffusion coefficient, which includes the effects of adsorption. Values of $D_w$ and $D_a$ reported or used by the Swedish repository program for bentonite are given in Table 2-4.

Table 2-4.  Diffusivity Values for Free Aqueous Solution and Saturated Bentonite.

| Element | $D_w \times 10^8$ (m$^2$ s$^{-1}$) [a] | $D_a \times 10^{12}$ (m$^2$ s$^{-1}$) Bentonite [b] |
|---|---|---|
| C | 1.2 | |
| Cl | 2 | |
| Cs | 2.1 | $2 - 5$ <br> $1.4$ [c] |
| I | 0.83 | $86 - 87$ |
| Ra | 0.89 | |
| Sr | 0.79 | $10.5 - 14$ |
| Th | 0.15 | |
| All other radionuclides | 1 | |

[a] SKB (2006b, Table A-40)
[b] SKB (2003a, Section 6.3)
[c] Ochs et al. (1998, Figure 1a)

## 2.5.5.  Radionuclide Solubility

An important control for the release and transport of radionuclides is aqueous solubility. PHREEQC version 2.14.2 and the thermo.com.V8.R6.230 database from Lawrence Livermore National Laboratories were used to estimate radionuclide solubility for Am, Np, Pu, Ra, Sb, Se, Sn, Tc, Th, and U for the general chemical conditions in granite at depth.

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[1] Other common formulations of this equation have a single variable $\tau$ or $\omega$ in place of the ratio $\frac{\delta}{\tau^2}$.

Based on the concentrations of Na, Ca, and Cl in the groundwater samples in Table 2-1, a brackish Na-Ca-Cl groundwater with a general recipe of 0.3 M NaCl, 0.05 M $CaCl_2$, and 0.001 M $Na_2SO_4$ was assumed in the calculations. Interaction with bentonite is not expected to alter this composition significantly except for some depletion of Ca due to ion exchange with Na (Posiva 2007, Section 3.2.2). Reducing conditions in this generic groundwater were defined based on the relative concentrations of sulfate and sulfide anticipated at the repository level. By fixing the hydrogen partial pressure at $10^{-7}$ atm (implying an equilibrium oxygen partial pressure of essentially zero [$10^{-69}$ atm at 25°C]), aqueous sulfate and sulfide concentrations were calculated within the ranges of data presented in Table 2-1. This constraint sets the Eh in the calculations to approximately -240 mV at pH 7.5 versus the standard hydrogen electrode. Because failure of the waste canister is generally expected to occur long after the thermal period, the temperature of the generic groundwater was set at 25°C. Ambient temperatures at repository depths at Olkiluoto, Forsmark, and Laxemar are between 10 and 15°C (Posiva 2010, Section 4.3; SKB 2006d, Table 9-4).

For radioelements whose solubility is likely limited by hydrolysis products, less soluble oxide phases were generally avoided in favor of more soluble hydroxide or hydrated phases. For uranium, uraninite ($UO_2$) was assumed to be the solubility-controlling phase because of its presence in the waste form. This treatment is consistent with a natural analogue study of uranium fixation in a Tertiary argillite (Havlova et al. 2006).

The pH of waters in granite systems tends to be neutral to slightly alkaline (Table 2-1). A bentonite buffer material surrounding the waste packages is not expected to significantly change the pH in the vicinity of the waste package. Bentonite contains calcite and a high density of surface exchange sites, each of which act to buffer pH in the slightly alkaline range (Posiva 2007, Section 3.2.2).

Results of the solubility calculations are shown in Table 2-5. These calculations assume a pH of 7.5. Higher pH values up to at least 8.5 reduce calculated solubilities or have no effect. The calculations at pH 7.5 are within a factor of 10, or greater than a factor of 10, of the values adopted in Posiva (2010, Table 1-9) (Sb was not evaluated in that report.). For the transport modeling in Section 4, Ac and Cm were assumed analogous to Am, and Pa was assumed analogous to Np. Solubility values for Nb, Pd, and Zr are from Posiva (2010, Table 1-9).

Because uraninite is the stable uranium phase in the waste form, and the solubility of uraninite under reducing conditions is low ($4 \times 10^{-10}$ M in this case), waste form degradation rates are also low under reducing conditions (on the order of $10^{-7}$ yr$^{-1}$; Posiva (2010, Section 1.4.6)). This implies that the release of actinides and fission products from the waste form may be so low that their aqueous concentrations may never be limited by solubility.

No limiting concentrations are presented in Table 2-5 for C, Cs, I, Sr, and Pb. The concentrations of these radioelements are expected to be controlled by their inventories, instant release fractions, and/or the slow dissolution rates of the waste forms (Section 4). Actual solubilities of these elements are moderate to high in this chemical environment.

Table 2-5. Radionuclide Solubilities Calculated at T = 25°C, pH 7.5.

| Radioelement | Solubility-Limiting Phase | Dissolved Concentration [a] (mol L$^{-1}$) | Notes |
|---|---|---|---|
| Am (Ac, Cm) | Am(OH)$_3$ | $6 \times 10^{-6}$ | Ac and Cm are assumed analogous to Am |
| Np (Pa) | Np(OH)$_4$ | $1 \times 10^{-9}$ | Pa is assumed analogous to Np |
| Nb | Nb(OH)$_5$ | $4 \times 10^{-5}$ | Posiva (2010, Table 1-9) |
| Pd | Pd(OH)$_2$ | $3 \times 10^{-6}$ | Posiva (2010, Table 1-9) |
| Pu | Pu(OH)$_4$ | $2 \times 10^{-7}$ | |
| Ra | RaSO$_4$ | $1 \times 10^{-6}$ | (SO$_4^{2-}$) fixed at $10^{-3}$ mol L$^{-1}$ |
| Sb | Sb(OH)$_3$ | $1 \times 10^{-7}$ | |
| Se | FeSe$_2$ | $4 \times 10^{-8}$ | |
| Sn | SnO$_2$ | $3 \times 10^{-8}$ | |
| Tc | TcO$_2$:2H$_2$O(am) | $3 \times 10^{-8}$ | |
| Th | Th(OH)$_4$ | $4 \times 10^{-7}$ | |
| U | UO$_2$ | $4 \times 10^{-10}$ | |
| Zr | Zr(OH)$_4$ | $2 \times 10^{-8}$ | Posiva (2010, Table 1-9) |

[a] Calculated using the PHREEQC code version 2.14.2 and the thermo.com.V8.R6.230 database from Lawrence Livermore National Laboratories, except where noted. The solution assumed 0.3 M NaCl, 0.05 M CaCl$_2$, $10^{-3}$ m Na$_2$SO$_4$, and $10^{-7}$ atm H$_2$ (g).

## 2.6. Chronology of Repository Conditions and Events

This section summarizes the expected evolution of the repository over one million years. The general framework of this section, and many of the noted events and expected behaviors, are taken from Gierszewski et al. (2004, Section 6.3). Their summary is taken from the description of the Base Scenario provided in McMurry et al. (2003), which includes justification, references and discussion of uncertainties. Additional information in the sections below originate from calculations specific to this report and from reports issued by the Swedish and Finnish programs.

### 2.6.1. 0 to 100 Years

Repository tunnels, rooms, and/or emplacement boreholes would be excavated during a period of approximately 30 years. An excavation damage zone (EDZ) would form within a few tens of centimeters of these excavations but would not form a continuous pathway for water movement (SKB 2006d, Section 9.2.2). Waste packages would be emplaced and surrounded by clay buffer. Emplacement tunnels or rooms would be backfilled and sealed, but access tunnels would remain

open. Monitoring would continue until repository closure at 100 years or beyond. During the period of operations and monitoring:

- Total radioactivity in individual waste packages would decrease as indicated in Figure 2-1 (UNF), Figure 2-2 (DOE HLW), and Figure 2-3 (reprocessing HLW). Waste package thermal power would decrease commensurate with decreases in radioactivity.

- Temperatures within the repository would peak (less than 100°C by design) and begin to decline (SKB 2006d, Section 9.3.4; Posiva 2006, Section 6.1.1).

- The initially low hydraulic pressures in the tunnels and clay buffer would induce water migration toward the waste packages. The clay buffer would swell as it hydrates.

- Minor corrosion of the copper canisters would occur due to the presence of oxygen. After backfilling, the oxygen would be limited; thus, copper corrosion by direct reaction with oxygen would be limited to a few tens of micrometers (SKB 2006d, Section 9.2.5; Posiva 2006, Section 6.4).

- Heat from the waste packages would warm the compacted clay buffer from the inside out. Any water vapor produced by evaporation near the waste package would condense further away where temperatures would be cooler. Hydration would thereby proceed inward.

- A small increase in alkalinity could result from groundwater interaction with cements and concrete, but the effects would be negligible (SKB 2006d, Section 9.2.7).

- Microbes would consume any stray organic materials, either aerobically in the presence of oxygen or perhaps by sulfate reduction (SKB 2006d, Section 9.2.7). Microbial activity would be inhibited within the hydrated clay buffer by lack of nutrients, swelling pressure, and the attendant small pore size.

## 2.6.2.  100 to 1,000 Years

At approximately 100 years, the repository would likely be closed and all access tunnels and shafts would be backfilled and sealed. Large thermal, hydraulic, chemical, and mechanical gradients would slowly dissipate as heat, water, and solutes disperse and swelling of the buffer helps to redistribute rock stresses.

- Temperatures in the repository would decline but remain elevated (SKB 2006d, Section 9.3.4).

- Air in the repository would be replaced by water.

- Oxygen in the pore space would be consumed by organic and inorganic processes. Reducing conditions would be established throughout the repository.

- Buffer material surrounding the waste packages and mixed into the backfill would swell as it hydrates, distributing loads across repository components and preventing spalling.

- Loads from the surrounding rock would be transmitted through the backfill and buffer onto the container. These loads would be high enough to compress the copper canister onto the steel inner vessel but too low to deform the inner vessel. They would also be high enough to eliminate microbes and prevent canister sinking (SKB 2006d, Section 9.3.9; Posiva 2006, Section 6.1.2).

- Reducing conditions would significantly limit corrosion rates.

## 2.6.3. 1,000 to 10,000 Years

The large gradients of the first one thousand years would continue to dissipate but more slowly as the geosphere absorbs and disperses the effects of the repository.

- Temperatures in the repository would slowly decline and level off at near ambient values near the end of this period (SKB 2006d, Section 9.3.4; Posiva 2006, Section 6.4).

- Cementitious seals and plugs would degrade allowing new flow pathways to develop (Posiva 2006, Section 6.4).

- The hydrated buffer material surrounding the waste packages would prevent significant advection of water, dissolved solutes, and colloids through the buffer.

- Corrosion of the copper canister would be limited by lack of oxygen and a very low flux of other reactants, most notably hydrogen sulfide, through the hydrated buffer to the waste package surface (SKB 2006d, Section 9.3.12).

- Reaction of groundwater with grout in fractures would raise the pH in these fractures to approximately 9 (SKB 2006d, Section 9.3.7).

## 2.6.4. 10,000 to 100,000 Years

From 10,000 to 100,000 years, the thermal pulse would be dispersed and temperatures within the repository (and within the waste packages) would nearly equal the temperature of the original host rock (SKB 2006d, Section 9.3.4). By the end of this time interval, the earth could be in the middle of the next glacial period. Depending on the latitude of the repository, permafrost followed by an ice sheet or glaciers could advance over the ground surface above the repository.

- Some replacement of sodium in the buffer material by calcium from groundwater and by dissolution of gypsum in the buffer would occur, but the extent would be limited and would not significantly affect buffer performance.

- Although permafrost could develop near the surface as the mean surface temperature decreases, freezing temperatures would not likely reach the repository (SKB 2006d, Section 9.4.3).

- Additional rock stresses, creep deformation, and fracturing around the repository due to the increased load of a potential glacier or ice sheet would not likely significantly affect waste package integrity or repository performance (SKB 2006d, Section 9.4.4).

- Corrosion of the copper outer barrier would continue to be limited and would not threaten containment as long as the buffer surrounding the waste package remained intact (SKB 2006d, Section 9.3.13).

- If the repository were located in the northern U.S. latitudes, groundwater flow rates at the repository level would decrease with the onset of permafrost at the ground surface due to decreasing infiltration. This effect could be counteracted by the presence of glaciers or a possible ice sheet. If the repository were located in the southern U.S., increased rainfall could increase the circulation of deep groundwaters.

## 2.6.5. 100,000 to 1,000,000 Years

Glacial periods could occur on a regular basis during this time interval. Based on ice core data, a glacial phase has occurred on average every 120,000 years over the past 650,000 years. However, major changes in ocean circulation, atmospheric composition, or the earth's crust could potentially disrupt the glacial cycle.

- Waste package thermal power would be insignificant in comparison to the geothermal flux.

- Retreating ice, which could occur multiple times during glacial periods, would enhance deep circulation of groundwater and could potentially cause the erosion of buffer materials surrounding waste packages.

- Large vertical movements in the host rock could occur as a result of ice loads, earthquakes, and slow crustal movements. The effects on rock mechanics would occur mostly at shallow depths, but glacial stresses would be lower than the stresses caused by excavation. Shear movements along existing fractures intersecting the repository could cause a small number of canister failures. The probability of such failures is expected to be low partly due to active avoidance of significant fractures during waste emplacement (Posiva 2006, Section 8.3.2).

- Erosion of buffer during periods of deep groundwater circulation could expose a small fraction of the waste packages to advective groundwater flow (SKB estimates this fraction to be 0.004 within the reference evolution of its license application (SKB 2011, Section 12.2)). This exposure would enhance corrosion rates by allowing a larger influx of corrosive reactants like hydrogen sulfide. In addition, erosion of the buffer would

allow sulfate-reducing bacteria to live near the waste package and generate hydrogen sulfide from an advective influx of sulfate.

- A small number of copper canisters exposed to advective flow (due to buffer erosion) could fail within one million years. Degradation of the canister insert and the waste form would begin soon thereafter, and radionuclides would be released from these waste packages.

# 3. SCENARIOS

A scenario in a repository safety analysis is a hypothetical combination of features, events, and processes (FEPs) acting upon the facility and its surroundings over time. A PA requires evaluation of a full set of reasonable scenarios to demonstrate the likely range of repository performance. Scenarios are developed by reviewing complete lists of FEPs and determining which FEPs could affect repository performance.

As described in Section 3.1, a FEP screening approach similar to that taken for both the Yucca Mountain repository and the WIPP is needed to identify potentially important FEPs to be included in a full PA. The current report does not provide a full PA, but it does discuss scenarios developed in international programs. Section 3.2 describes the scenarios included in the safety analysis in Section 4 and how the FEPs were used to define the scenarios.

## 3.1.  Identification of Relevant FEPs

Developing relevant scenarios for a PA involves five basic steps (DOE 2008; DOE 1996): (1) identification of a list of FEPs potentially relevant to long-term performance of the disposal system; (2) evaluation of FEPs from this list to determine which to include in the PA and those to omit; (3) construction of scenarios from included FEPs for further screening or analysis; (4) selection of scenarios to include in the PA; and (5) implementation and analysis of the scenarios in the PA. Consistent with the approach taken in 40 CFR 197, it is assumed that the mean annual dose from a repository in granite will include probability-weighted consequences of releases due to all significant FEPs and will account for uncertainty associated with those FEPs.

Various programs in the United States and other nations have compiled exhaustive lists of FEPs for mined geologic disposal of radioactive wastes. For this analysis, the FEP list developed for the DOE Used Fuel Disposition Campaign (Freeze et al. 2010) was used. This list, developed from international FEP lists, currently includes 208 FEPs potentially relevant to a wide range of disposal system alternatives.

Each FEP is evaluated against screening criteria provided in U.S. regulations. Specifically, EPA standard 40 CFR 197 states that FEPs that have a probability of occurrence lower than 1 in 10,000 over 10,000 years (or an annual probability of occurrence less than one in $10^8$ during the first 10,000 years after closure) may be excluded from the analysis. FEPs that have higher probabilities, but do not significantly change the results of long-term PAs, may also be omitted (40 CFR 197.36(a)(1)). In addition, some potentially relevant FEPs are screened from further consideration because they are inconsistent with specific aspects of the regulatory requirements. For example, existing regulations for the WIPP and the Yucca Mountain repository indicate that PAs should not include consequences of deliberate human acts of sabotage or disruption in the far future.

Analyses developed for granite repository programs are useful when deciding whether to include or exclude a FEP. Exclusion of criticality FEPs, for example, is consistent with conclusions drawn by SKB (2006d, Section 10.3). In addition, analyses developed for other types of repositories may be useful, as is the case with arguments presented in Brady et al. (2009, Section 4.3.1) justifying the exclusion of criticality FEPs for deep borehole disposal.

Each of the 208 FEPs developed in the DOE Used Fuel Disposition Campaign has been considered (screened) for potential relevance to disposal in granite formations. Table B-1 in Appendix B summarizes the likely screening decisions for each FEP (whether a FEP is likely to be included in, or excluded from, a full PA for a granite repository).

## 3.2. Scenario Selection

Scenarios evaluated in international granite repository PAs typically include a main scenario, less probable scenarios, and variants of these scenarios. In the main scenario of a recent case study of the Canadian safety assessment, there are no waste package failures during the performance period because the canisters and repository perform as designed (Gierszewski et al. 2004, Section 6). In the main scenario of the Swedish PA, waste packages only become breached when the buffer fails to prevent advective conditions between the waste package and host rock (SKB 2006d, Section 12.2.2). In these cases, the buffer erodes for a small fraction of waste packages (< 0.1) due to deep penetration of dilute glacial groundwater (SKB 2006d, Section 12.3.1). Less probable scenarios considered and simulated in Finnish, Swedish, and Canadian programs include

- defective waste package,
- canister failure due to shear movement,
- buffer emplacement mishaps,
- human intrusion, and
- gas transport of radionuclides.

Scenarios ruled to be of low consequence or to have negligible probability based on logical arguments and independent analyses in the Finnish, Swedish, and Canadian programs include

- canister failure due to isostatic load from independent or combined forces, including buffer swelling and glacial overburden (SKB 2006d, Section 12.8.5),
- buffer freezing, and
- oxygen penetration to the waste package (SKB 2006d, Section 12.11.1).

In a full PA, many processes that might otherwise constitute a scenario are either implicitly included in the dose calculations or explicitly excluded. For example, flow along the EDZ or through sealed shafts and backfilled tunnels (e.g., FEPs 2.1.08.03, 2.1.09.51, and 2.2.08.06 in Appendix B) may not be specified as scenarios in a full PA because two- and three-dimensional models that include the properties of the EDZ, shafts, and tunnels will include these processes and show whether flow along these pathways is significant. Alternatively, independent calculations may show that these FEPs can be excluded from the model if the calculations indicate they have negligible effects.

Climate scenarios are typically superimposed on the various scenarios modeled. In the Finnish models, for example, glacial periods are defined and estimates of anthropogenic greenhouse gases are considered (Posiva 2010, Section 3.1). These scenarios have the potential to lead to

changes in groundwater circulation, flow rates, and composition and are included in the model calculations to ensure that potential climate changes are considered in the safety analysis.

For the purposes of the safety analysis in this study, the following assumptions are made regarding the repository environment and potential exposure pathway:

- The repository is placed in granite where there is a low hydraulic gradient.

- The repository is located deep in the saturated zone where reducing conditions persist even during periods of deep penetration of glacial melt water.

- A future groundwater well is constructed at a distance of 500 m down-gradient of the repository.

- Migration of radionuclides along the EDZ and through tunnels and shafts is assumed to be insignificant due to effective backfilling and sealing. The EDZ is not expected to form a continuous conductive flow path (SKB 2006d, Section 9.3.14) and backfill is expected to have lower hydraulic conductivity than the surrounding rock (Section 2.4.2).

- Though a release scenario from a human intrusion disruptive event is discussed below, it is not considered in the safety analysis in Section 4.

Based on the FEP screening in Appendix B, the scenarios of interest in international programs, and the assumptions above, five scenarios are identified and examined in more detail. Each is addressed in the following paragraphs. Two of these scenarios, the defective waste package scenario and the buffer failure scenario, are simulated in Section 4.

**Nominal Scenario: The waste packages and EBS perform as designed.** *No releases occur because the waste form remains contained within the waste package during the entire performance period.* This scenario maintains that the EBS will protect the waste packages from significant damage during the performance period, preventing canister failure and radionuclide release. This scenario is consistent with the base scenario of the third case study for the Canadian concept (Gierszewski et al. 2004) and the expected performance of the Swedish repository, barring an unexpectedly large earthquake or significant buffer erosion (SKB 2006d, Section 9.5.1).

**Defective Waste Package Scenario: A major defect in a waste package allows early radionuclide release.** *A waste package is assumed to have a major defect at the time of emplacement.* This scenario assumes that one waste package in the capture zone of the future groundwater well has a major defect. One waste package is selected instead of multiple waste packages based on estimates of undetected defect rates and the number of waste packages in the capture zone (Section 4.1). The defect is simulated by assuming the waste package provides no barrier performance. Exposing the full inventory of a defective waste package is a pessimistic assumption because undetected defects would likely be small and the waste package would still provide some performance. In addition, for UNF, early and complete failure of all cladding in the defective waste package is extremely unlikely. Nevertheless, at the time of repository closure it is assumed that the entire waste form is exposed to water and begins to degrade. At this early time, radiolytic phenomena may increase waste form degradation rates above solubility limited rates; however, this increase is assumed to be captured by the conservative treatment of waste

form exposure. The buffer, backfill, and seals perform as designed, causing the primary pathway to be through the geosphere. Released radionuclides diffuse through the bentonite buffer and migrate to the well via a near-field fracture and a far-field fracture zone.

**Buffer Failure Scenario: Deep circulation of glacial melt waters causes buffer erosion.** *Corrosion of a number of waste canisters is enhanced by the erosion of buffer materials caused by hydrologic changes brought on by the next glacial cycle.* This scenario assumes that the earth's glacial cycle continues such that an ice sheet or glacier advances over the top of the repository site and then, at approximately 100,000 years in the future, retreats during a subsequent warming period. The warming period is assumed to cause deep penetration of melt waters at the repository site as the ice retreats. Similar to the probability distributions presented by SKB (2006d, Figure 9-99), the increased flow conditions at depth are assumed to last approximately 25,000 years and to sufficiently erode the buffer to expose one quarter of the waste packages to advective groundwater flow. The increased corrosion rate due to flowing groundwater causes a small fraction of these waste packages to fail within one million years. Internal waste package components, such as the insert and fuel cladding, are pessimistically assumed to fail when the canister fails, initiating degradation of all waste in the failed packages. The backfill and seals may be affected but are assumed to perform as intended. However, for emplacement boreholes containing failed waste packages, the buffer is assumed to be completely gone. Thus, radionuclides released from breached waste packages are assumed to migrate directly to the host rock.

**Shear Movement Scenario.** *An earthquake causes a displacement that ruptures waste packages.* This scenario is not simulated in the generic safety assessment because it is assumed that the repository will be located in granite that has a low probability of significant earthquakes and of major glacially induced faulting. Site-specific calculations would be needed to confirm exclusion of this scenario. Calculations at Forsmark and Laxemar that combine the effects of isostatic pressure from the ice load of a glacial period and the probabilities for canisters being damaged within one million years by earthquakes give a mean canister failure rate of less than $2 \times 10^{-5}$, which is less than one in 50,000 canisters (SKB 2006d, Section 9.5.1).

**Disruptive, Human Intrusion Scenario.** *A borehole is drilled through the repository and later abandoned; a vertical hydrologic gradient transports radionuclides to a shallow aquifer from which they are pumped to the biosphere.* This is a stylized calculation specified by 40 CFR 197. Implementation for a granite repository would be inherently similar to the human intrusion scenario in the PA for the WIPP (DOE 2009) but would not be as likely to happen in granite and is therefore not analyzed in this report.

# 4. PERFORMANCE ANALYSIS

Generic performance analyses were performed for the defective waste package and buffer failure scenarios described in Section 3.2. The conceptual model for each of these scenarios assumes that released radionuclides migrate through the granite to a domestic groundwater supply well, as illustrated in Figure 4-1.
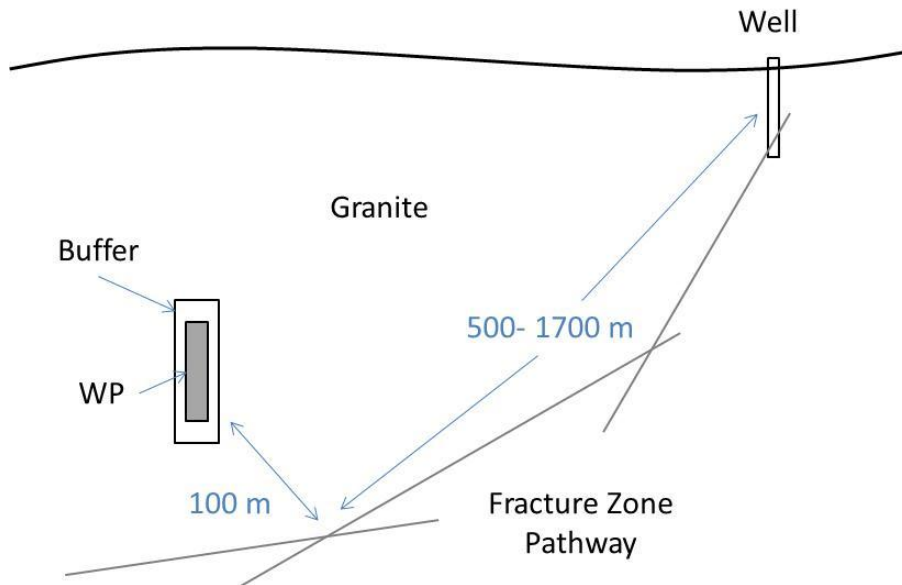


Figure 4-1.  Conceptual Model of the Nominal Radionuclide Pathway.

## 4.1.  Model Design

The model boundary conditions and assumptions include the following:

- The repository is located at a depth of 500 m in sparsely fractured granite. The layout chosen for these calculations is the vertical emplacement design shown on the left side of Figure 2-4. The emplacement spacing is that of the Forsmark design. In that design, emplacement tunnels are 40 m apart and the spacing between emplacement boreholes is 6 m. Because prevention of temperatures (>100°C) that could damage the buffer is a top priority in the design of the repository, waste package spacing at a selected location would depend largely on the heat output of the waste packages and the heat conductivity of the host rock.

- The total inventory, assuming UNF is reprocessed, consists of 32,154 waste packages for UNF, 5,003 waste packages for HLW glass, and 4,055 waste packages for RW (Appendix A). However, only UNF waste packages are considered in the safety analysis and only 3,000 of them are assumed to be emplaced within the capture zone of the well

(This assumption is discussed in Section 4.3.). Each waste package has a copper canister with 50-mm wall thickness. The initial radionuclide inventory is consistent with Appendix A. Decay and in-growth are included in the calculations.

- Corrosion rates for the copper canisters in the buffer failure scenario are sampled from a cumulative probability distribution applicable to advective conditions (Table 4-1). Corrosion rates for canisters with intact buffer are too low to cause canister failure within one million years. In the defective waste package and buffer erosion scenarios the canister contents, such as the cladding surrounding the UNF, provide no further performance upon canister containment failure and are effectively treated thereafter as nonexistent.

- Once the canister is breached, the release of radionuclides from each waste package is limited by

  - waste form degradation rates that initiate upon waste canister breach, except for instant release fractions (Table 4-1);
  - radionuclide solubility (Table 2-5); and
  - radionuclide diffusion into the buffer surrounding the waste packages when the bentonite buffer is intact; otherwise (in the buffer failure scenario), release is limited by the flux of water flowing through fractures intersecting the waste package borehole.

- Radionuclide source term concentrations are further limited by the isotope mole fraction for radioactive elements that have more than one isotope present in the waste form. Assuming congruent release of the isotopes, the maximum aqueous concentration for each isotope is limited to the elemental solubility limit multiplied by the mole fraction of that isotope.

- Diffusion is modeled through 0.35 m of bentonite buffer. Diffusion is limited by adsorption and pore geometry (Section 2.5.4). In the buffer failure scenario, diffusion through the buffer and adsorption to the buffer are not modeled because the buffer is assumed to have eroded away.

- Flow through 100 m of near-field sparsely fractured granite is represented by a dual porosity fracture flow model where advection occurs in the fractures and diffusion and adsorption occur in the diffusion porosity (Section 2.5). Flow through the near field is modeled as flow through a block of sparsely fractured granite with general dimensions as shown in Figure 4-2. The model does not require the near-field fracture to be planar or to intersect the borehole at a specific angle. Figure 4-3 illustrates the advective flow modeled through a near-field fracture and EDZ and around intact buffer materials.

Table 4-1.  Near Field Model Parameters and Parameter Values.

| Near Field Parameter | Value | Supporting Information |
|---|---|---|
| Number of UNF waste packages (WPs) | 3,000 | Approximately 9% of UNF inventory (Appendix A), consistent with simulated capture zone of assumed water supply well (Section 4.3) |
| WP length (m) | 5 | Figure 2-6; Posiva (2010, Section 4.5.1) |
| WP diameter (m) | 1 | Figure 2-6; Posiva (2010, Section 4.5.1) |
| WP copper canister wall thickness (m) | 0.05 | Figure 2-6; Posiva (2010, Section 4.5.1) |
| WP interior void fraction | 0.4 | Approximated for a degraded WP |
| WP undetected defect rate for copper canister | $2 \times 10^{-4}$ | Gierszewski et al. (2004, p. 39) |
| WP copper corrosion rate under diffusion conditions (mm yr$^{-1}$) | $5 \times 10^{-6}$ | Posiva (2010, Section 2.3.1) |
| WP copper corrosion rate under advective conditions, i.e., after glacial melt waters erode buffer around WP (mm yr$^{-1}$) | $1 \times 10^{-6}$ (0.80) <br> $3 \times 10^{-6}$ (0.90) <br> $1 \times 10^{-5}$ (0.95) <br> $5 \times 10^{-5}$ (0.99) <br> $7 \times 10^{-5}$ (0.9938) <br> $1 \times 10^{-4}$ (0.996) <br> $2.5 \times 10^{-4}$ (0.9991) <br> $3.5 \times 10^{-4}$ (0.9997) <br> $5 \times 10^{-4}$ (1.00) | Cumulative probability distribution based on copper corrosion rates estimated for Laxemar for advective conditions (SKB 2006d, Figure 9-102) (Note that corrosion rates greater than $5 \times 10^{-5}$ mm yr$^{-1}$ are needed to penetrate the 50 mm copper shell within one million years.) |
| Instant release fractions for UNF | 0.0001 (Sn) <br> 0.001 (Se) <br> 0.01 (Sr, Tc, Pd) <br> 0.05 (Cs, I) <br> 0.1 (C, Cl) | Posiva (2010, Table 1-5) |
| Fractional UNF degradation rate (yr$^{-1}$) | $1 \times 10^{-8}$ (min) <br> $1 \times 10^{-7}$ (mode) <br> $1 \times 10^{-6}$ (max) | SKB (2006d, Table 10-3) <br> Log-uniform distribution used in buffer failure scenario; mode used in defective waste package scenario |
| WP radionuclide inventory | tabulated | Appendix A |
| Radionuclide solubility in EBS | tabulated | Table 2-5 |
| Bentonite buffer thickness around WPs (m) | 0.35 | Posiva (2010, Figure 4-9); SKB (2006d, Figure 4-4) |
| Bentonite porosity | 0.44 | Posiva (2010, Section 3.2.3) |
| Bentonite dry bulk density (kg m$^{-3}$) | 1600 | Posiva (2010, Section 3.2.4) |
| Bentonite tortuosity ($\tau$) (Eq. 2-5) | 4 | Ochs et al. (1998, Table 3) |
| Bentonite constrictivity (Eq. 2-5) | 1 | Assumed, as in Ochs et al. (1998) |
| Bentonite adsorption distribution coefficients | tabulated | Table 2-2 |

Table 4-1.  Near Field Model Parameters and Parameter Values. (Continued)

| Near Field Parameter | Value | Supporting Information |
|---|---|---|
| Diffusivity in aqueous solution | tabulated | Table 2-4 |
| Near field granite path length (m) | 100 | Assumption based on avoidance of waste emplacement near fracture zone |
| Near field granite flow porosity | 0.0001 | SKB (2006b, p. 159); Posiva (2008a, pp. 17-18); Five times higher than Posiva (2009, Table 3-3) |
| Near field granite diffusion porosity | 0.001 | 10 times lower than that used by Posiva (2009, p. 29), in the sparsely fractured granite; Posiva (2008a, p. 19) |
| Near field granite dry bulk density (kg m$^{-3}$) | 2,650 | Carmichael (1982); Lama and Vutkuri (1978) |
| Near field granite matrix tortuosity ($\tau$) (Eq. 2-5) | 3.2 | Approximated from $\tau^{-2} = (porosity)^{1/3}$ |
| Near field granite pore velocity (m yr$^{-1}$) | 0.01 | Implies Darcy velocity of $3 \times 10^{-13}$ m s$^{-1}$, which is in the middle of the range of Posiva (2009, p. 80) at repository level |
| Near field granite fracture velocity (m yr$^{-1}$) | 0.1 | Ratio of flow porosity to diffusion porosity |
| Near field granite conduit bulk cross-sectional area (m$^2$) | 10 | 2 m $\times$ 5 m |
| Near field granite mean fracture aperture (m) | $5 \times 10^{-4}$ | Posiva (2010, p. 279) |
| Near field granite length between fractures (m) | 5 | Ratio of fracture aperture to flow porosity |
| Near field granite longitudinal dispersivity (m) | 10 | Assumed 10% of path length |
| Near field granite matrix adsorption distribution coefficients | tabulated | Table 2-3 |
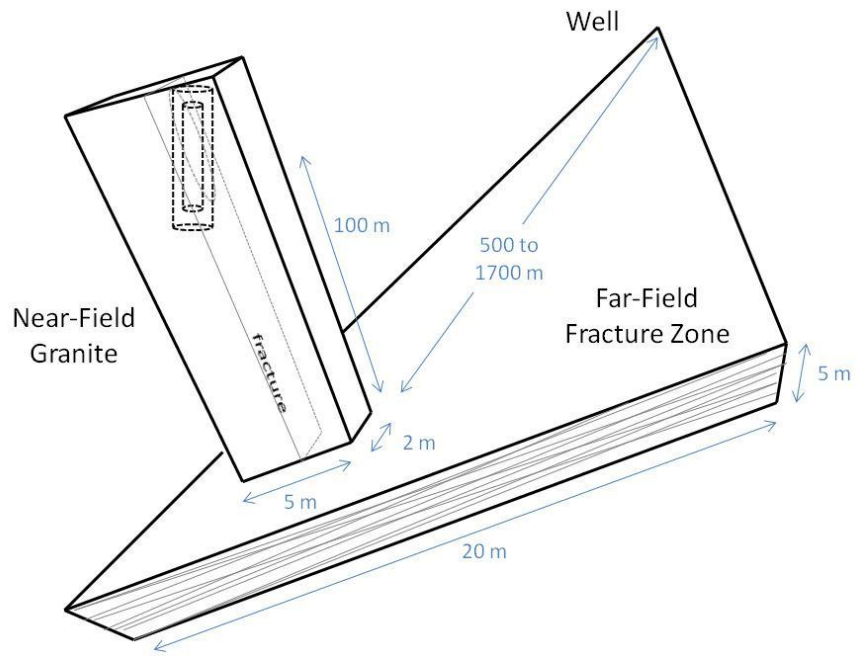
Figure 4-2.  Flow Paths and Dimensions.
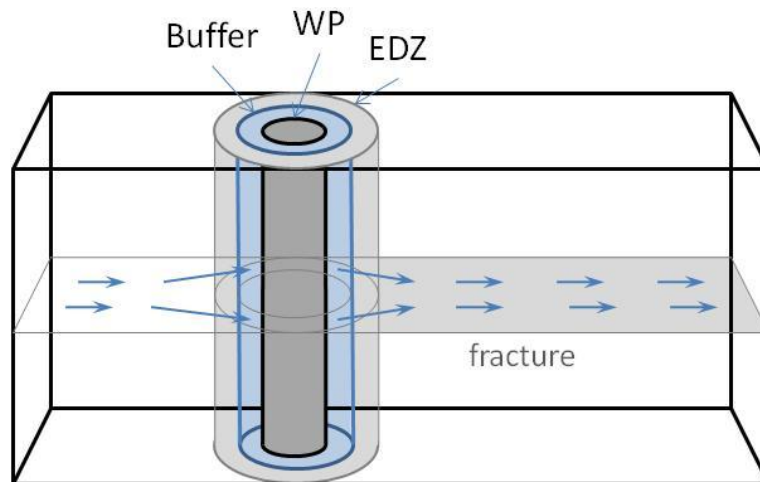


Figure 4-3.  Simulated Near Field Barriers and Flow Paths.

- Flow through 500 m to 1,700 m of a hydraulically conductive fracture zone in the granite (Figure 4-2) is represented by a dual porosity fracture flow model where advection occurs in the fractures, and diffusion and adsorption occur in the diffusion porosity. Flow through the fracture zone is modeled as flow through a block of highly fractured granite. This block is divided into conduits, each of which captures flow from 100 waste packages.

- The receptor well is assumed to capture flow from fractures intersecting the emplacement boreholes of 3,000 waste packages (i.e., 30 fracture zone conduits). Based on the spacing of the waste packages and emplacement drifts, 3,000 waste packages would cover an area of at least 0.72 km$^2$. A highly simplified model representation of the 3,000 waste packages within the capture zone of the well is shown in Figure 4-4. The decision to limit the number of waste packages within the capture zone to 3,000 is discussed in Section 4.3.

- Flow velocities, flow porosities, diffusion porosities, fracture spacing, and fracture apertures in the near field granite and fracture zones are largely borrowed from data and modeling results from international programs (Table 4-1 and Table 4-2).

- Colloids are not modeled in the calculations. A clay buffer will physically prevent colloid migration, and the high ionic strength of groundwater at repository depth will chemically limit colloid stability (Buck and Sassani 2007). In the buffer failure scenario, deep circulation of dilute glacial melt water at 100,000 years is assumed to stabilize colloids for a period of approximately 25,000 years and cause the buffer around many waste packages to erode away. Waste packages with eroded buffer are nevertheless expected to remain intact long after ionic strength is assumed to return to levels that inhibit colloid stability. Although deep penetration of glacial melt waters may be insignificant or unlikely, based on the isotope data at Forsmark and Olkiluoto (Section 2.2.2), a rigorous site-specific PA should include a careful evaluation of the potential for significant colloidal processes.

- The receptor is assumed to be a self-sufficient farming family that uses water from a well drilled into the granite within 500 m of the repository. The well pumps at a rate of 738 m$^3$ yr$^{-1}$, which is on par with the average water use of a family of six (Van der Leeden et al. 1990). The implications of the selection of the well discharge rate and the number of people exposed are discussed in Section 4.3.

- The IAEA BIOMASS Example Reference Biosphere 1B (ERB1B) dose model is used to calculate the annual dose based on well water consumption (IAEA 2003). In the analysis, the well is assumed to capture all flow from the simulated fractures down-gradient of the 3,000 waste packages.

Plan View (partial repository)
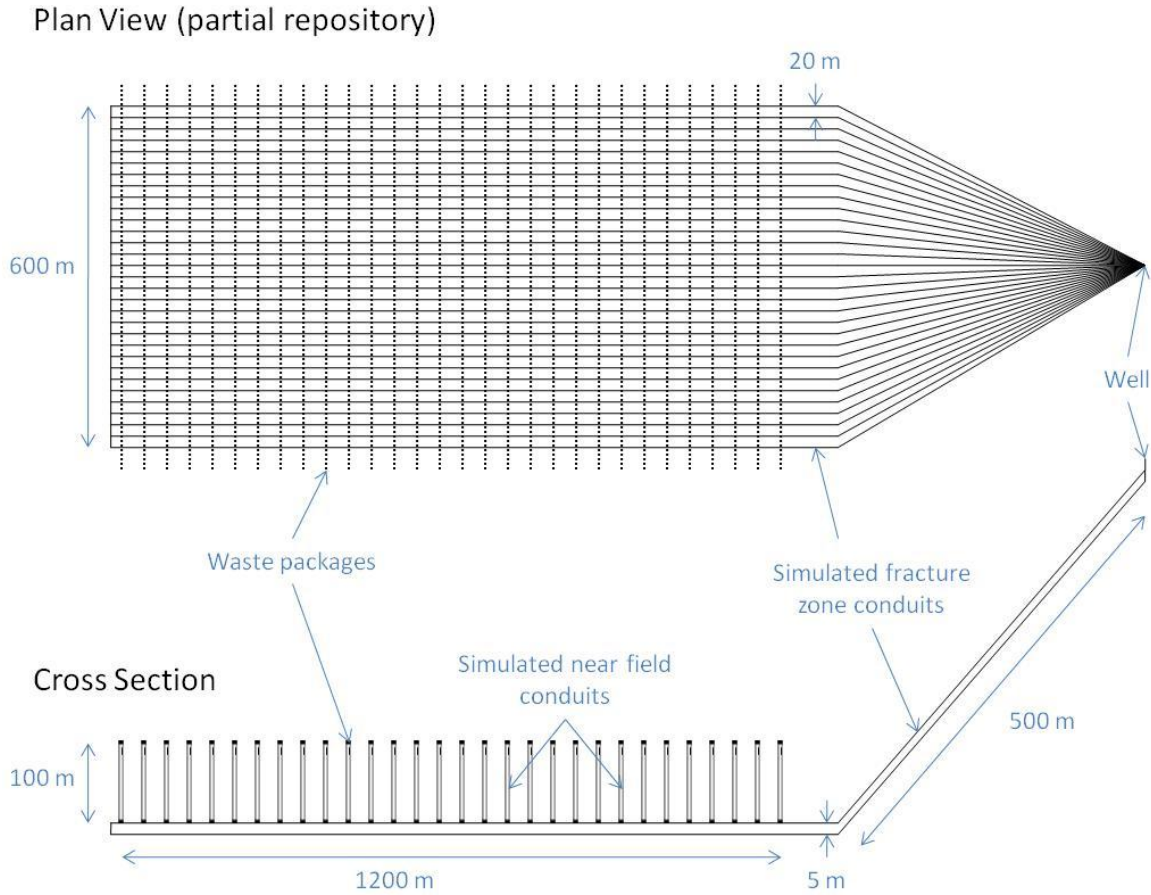


Figure 4-4.  Simplified Conceptual Model Layout of 30 Fracture Zone Conduits Capturing Flow from the Boreholes of 3,000 Waste Packages.

- For the buffer failure scenario

    - 25% of the waste packages up gradient of the well are assumed to become exposed to advective groundwater flow at 100,000 years (This fraction is within the range estimated by SKB (2006d, Figure 9-99) assuming spalling but is far above the estimated range assuming no spalling ($<3\%$));

    - a probability distribution of higher copper canister corrosion rates is used for waste canisters exposed to advective groundwater flow (Table 4-1); and

    - canister contents including fuel cladding are assumed to provide no barrier capability after failure of the canister.

Table 4-2. Far Field Model Parameters and Parameter Values.

| Far Field Parameter | Value | Supporting Information |
|---|---|---|
| Fracture zone path length (m) | 500 - 1700 | Uniform distribution. This distribution represents the approximate distance between a shallow well and the WPs within the capture zone of the well, after subtracting the near field distance between the fracture zone and the WP. For the defective waste package scenario, this value is set at 500 m. |
| Fracture zone flow porosity | 0.002 | SKB (2003b, p. 95); Posiva (2009, Table 3-2) |
| Fracture zone pore velocity (m yr$^{-1}$) | 0.01 | Approximately ten times higher than the cumulative pore velocity through the near field and far field in Gierszewski et al. (2004, p. 45); implies Darcy velocity of $3 \times 10^{-12}$ m s$^{-1}$, which is on the low end of the range of Posiva (2009, p. 80) for hydraulically conductive zones at Olkiluoto. |
| Fracture zone fracture velocity (m yr$^{-1}$) | 0.05 | Ratio of flow porosity to diffusion porosity |
| Fracture zone conduit bulk cross-sectional area (m$^2$) | 100 | 5 m $\times$ 20 m |
| Fracture zone dry bulk density (kg m$^{-3}$) | 2,650 | Assumed equivalent to near field granite |
| Fracture zone mean fracture aperture (m) | $5 \times 10^{-4}$ | Posiva (2010, p. 279) |
| Fracture zone length between fractures (m) | 0.25 | Ratio of fracture aperture to flow porosity |
| Fracture zone longitudinal dispersivity (m) | 50 - 170 | Assumed to be 10% of path length |
| Fracture zone diffusion porosity | 0.01 | Posiva (2009, p. 29); Posiva (2008a, p. 19) |
| Fracture zone matrix tortuosity ($\tau$) (Eq. 2-5) | 2.2 | Approximated from $\tau^{-2} = (\text{porosity})^{1/3}$ |
| Fracture zone matrix adsorption distribution coefficients | tabulated | Table 2-3 |
| Number of WPs up gradient of each fracture zone conduit | 100 | Flow through each near field conduit is 1% of the flow through the fracture zone conduit |
| Number of fracture zone conduits up gradient of well | 30 | Implies 3,000 WPs up-gradient of well; also implies 0.6 WP with undetected defect in copper wall up-gradient of well (round up to 1 WP) |
| Well pump rate (m$^3$ yr$^{-1}$) | 738 | Value borrowed from Gierszewski et al. (2004) to make generally consistent with capture zone dimensions and path lengths. Equivalent to the use rate of a family of six, based on average 86.5 gal person$^{-1}$ day$^{-1}$ (Van der Leeden et al. 1990). |
| Granite heat conductivity (W m$^{-1}$ K$^{-1}$) | 3 | Approximated based on values at Forsmark and Laxemar (SKB 2006d, Table 9-4) |
| Granite heat capacity (MJ m$^{-3}$ K$^{-1}$) | 2.2 | Approximated based on values at Forsmark and Laxemar (SKB 2006d, Table 9-4) |

## 4.2. Model Results

Performance analysis of a generic granite repository was conducted for two scenarios, a defective waste package scenario and a buffer failure scenario. The defective waste package scenario analysis was deterministic and the buffer failure scenario analysis was probabilistic. Each simulation was performed using GoldSim version 10.5.

### 4.2.1. Defective Waste Package Scenario

As discussed previously, this scenario assumes that one waste package has undetected major defects and fails at the time of repository closure, exposing the full radionuclide inventory of the waste package. The waste package is assumed to contain commercial UNF, and the potential performance of the fuel cladding is not considered. For this scenario we assume that the bentonite buffer, backfill, and seals perform as designed and the primary transport pathway for released radionuclides is through the geosphere. Released radionuclides diffuse through the bentonite buffer and migrate to the well via a near-field fracture and a far-field fracture zone.

Figure 4-5 shows the mass flux of radionuclides from the failed waste package to the buffer. Because potential performance of the failed waste package as a barrier to radionuclide transport is not considered, radionuclides released from the UNF are assumed to directly contact the entire inner surface of the buffer layer (35 cm thick) and are modeled to migrate through the buffer layer by radial diffusion. The initial high release rates of $^{129}$I and $^{107}$Pd within the first 30 years are due in part to initial instantaneous release from the waste form (Table 4-1). Much of the instantly released $^{107}$Pd precipitates within the waste package, limiting release to the buffer to a rate of 0.6 g yr$^{-1}$ for 20 years while the precipitated mass fully dissolves. Other radioelements precipitating in the waste package include uranium, technetium (fully dissolved at 13,700 years), neptunium, and zirconium (after 100,000 years). Between 100 and 10,000 years, the radionuclides with high release rates are $^{239}$Pu, $^{99}$Tc, $^{240}$Pu, $^{135}$Cs, $^{93}$Zr, and $^{241}$Am. At one million years, the dominant release rates are for $^{93}$Nb, $^{229}$Th, $^{232}$Th, $^{93}$Zr, and $^{135}$Cs. Note that the radionuclide release rates, especially for radionuclides with long half-lives, show persistent, relatively steady release behaviors until their decay effect becomes dominant. This behavior is largely due to a low fractional waste form degradation rate of $10^{-7}$ yr$^{-1}$, which is equivalent to a half-life of approximately seven million years. After release of the instantly-released radionuclides, the $10^{-7}$ yr$^{-1}$ rate implies that the mass flux from the waste form (g yr$^{-1}$) decreases only by 1% over 100,000 years and by 10% over one million years.

The radionuclide mass flux from the bentonite buffer is shown in Figure 4-6. It is assumed that the buffer layer maintains its physical configuration and integrity for the entire simulation period. Cesium, strontium, and iodine are the first radioelements to break through at high rates due to their instantaneous release fractions (Table 4-1) and low adsorption (Table 2-2). A comparison of Figure 4-6 and Figure 4-5 shows that the breakthrough curves of radioelements that adsorb strongly to the buffer are considerably delayed. (Based on these figures, the peak release rate of $^{135}$Cs from the buffer appears to be higher than any release rates from the waste package. This is not the case. The peaks from the waste package involving instantaneously-released radionuclides break through largely or entirely within the first 10 years. For example, at approximately 0.2 yr, the peak waste package release rate of $^{135}$Cs exceeds 200 g yr$^{-1}$.)
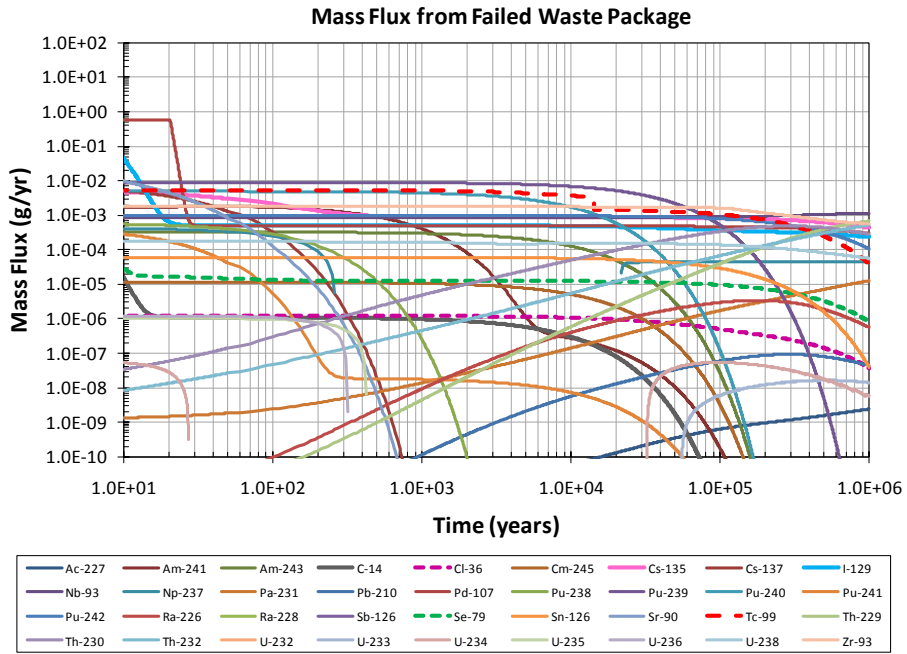
Figure 4-5.  Radionuclide Mass Flux from the Breached Waste Package in the Defective Waste Package Scenario.
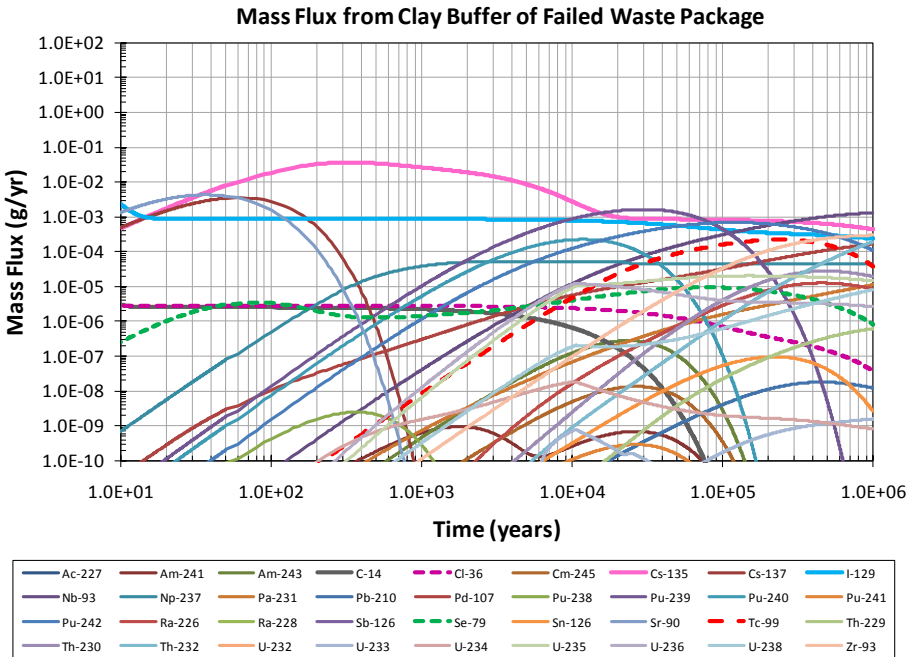


Figure 4-6.  Radionuclide Mass Flux from the Buffer of the Breached Waste Package in the Defective Waste Package Scenario.

Figure 4-7 shows the mass flux of radionuclides from the near-field fracture in the defective waste package scenario. As shown in the figure, only three radionuclides, which are characterized as mobile, non-sorbing, and highly soluble ($^{129}$I, $^{36}$Cl and $^{14}$C), are released from the near-field fracture at appreciable rates. All other radionuclides are amply retarded by adsorption. The $^{129}$I peak release rate is about $7.4 \times 10^{-4}$ g yr$^{-1}$ at 23,000 years and is about two orders of magnitude greater than the next dominant radionuclide peak release ($^{36}$Cl peak flux of $2.1 \times 10^{-6}$ g yr$^{-1}$ at 21,000 years). A pulse of $^{129}$I released from the near-field fracture is shown using a linear scale in Figure 4-8. The release rates of these radionuclides following their peaks steadily decrease for the rest of the analysis period due largely to radioactive decay.



**Mass Flux from Near-Field Fracture Zone**
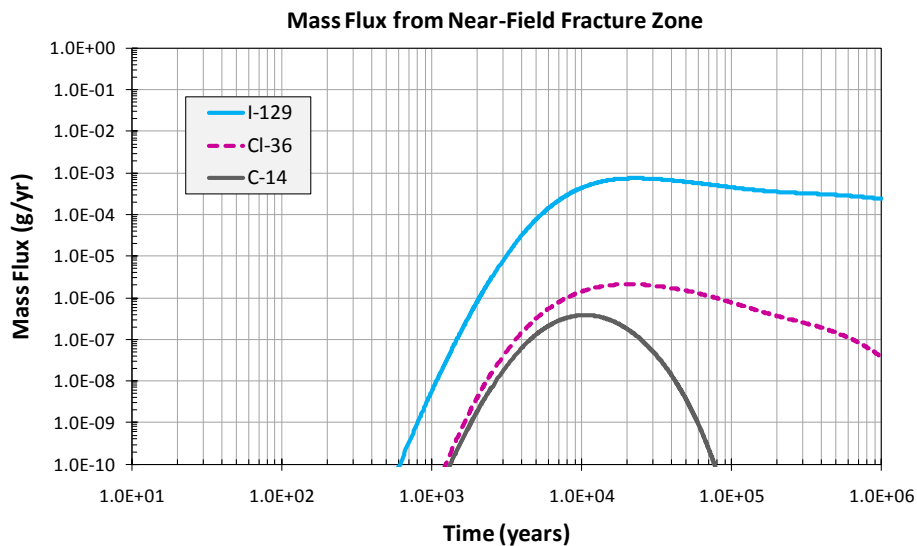
Figure 4-7.  Radionuclide Mass Flux from the Near-Field Fracture in the Defective Waste Package Scenario.

Radionuclide release rates from the far-field fracture zone are shown in Figure 4-9. $^{129}$I is the dominant released radionuclide. The peak release rate ($4.4 \times 10^{-4}$ g yr$^{-1}$) is about a factor of two lower than the near-field peak release rate, and the time for the peak is delayed to about 180,000 years (vs. 23,000 years for the near-field release peak). Similar effects are calculated for $^{36}$Cl (peak release rate of $7.9 \times 10^{-7}$ g yr$^{-1}$ at 145,000 years compared to $2.1 \times 10^{-6}$ g yr$^{-1}$ at 21,000 years for the near-field release). The far-field release rate for $^{14}$C is negligibly small.

Figure 4-10 shows the radionuclide dose rate at the hypothetical accessible environment. $^{129}$I is the dominant radionuclide for dose, and the peak dose rate is calculated to be about $4.8 \times 10^{-2}$ mrem yr$^{-1}$ at 180,000 years. The dose rates from $^{36}$Cl and $^{14}$C are much smaller.
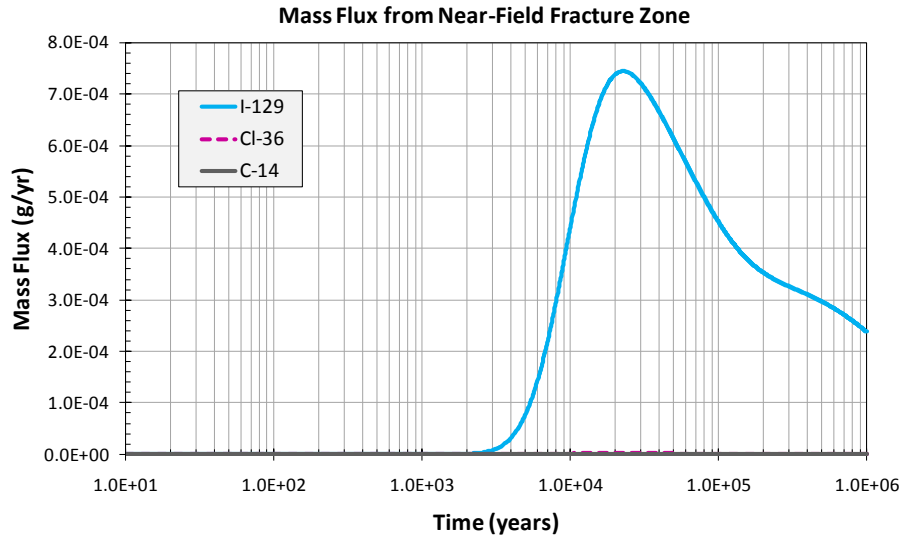
Figure 4-8. Radionuclide Mass Flux from the Near-Field Fracture in the Defective Waste Package Scenario (linear scale for mass flux).
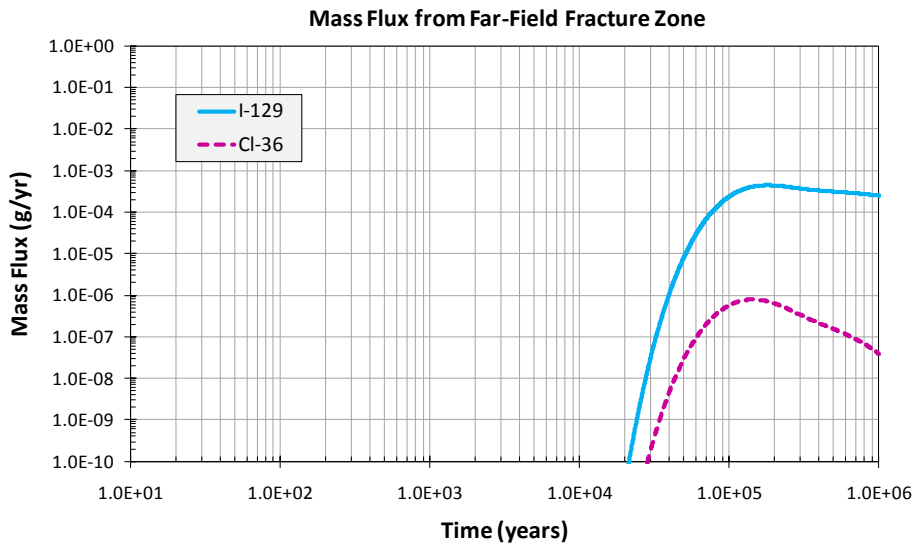


Figure 4-9. Radionuclide Mass Flux from the Far-Field Fracture Zone in the Defective Waste Package Scenario.

**Dose at Hypothetical Acessible Environment**
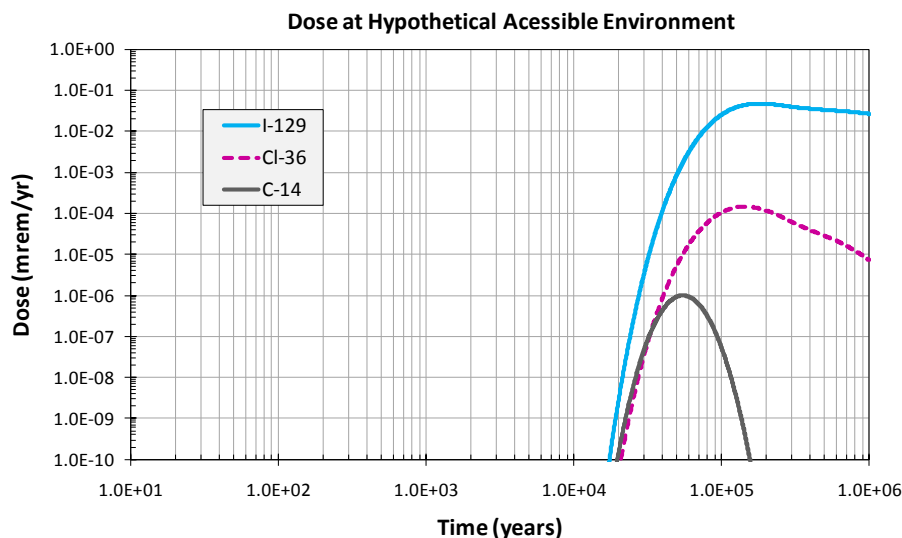
Figure 4-10.  Radionuclide Dose Rate at the Hypothetical Accessible Environment for the Defective Waste Package Scenario.

### 4.2.2.  Buffer Failure Scenario

The model for the buffer failure scenario assumes a total of 3,000 waste packages in the capture zone and 30 fracture zone conduits down gradient of these waste packages (Figure 4-2 and Figure 4-4). The analysis for this scenario was conducted probabilistically with a total of 1,000 realizations. Three model parameters were sampled randomly: 1) the waste canister corrosion rate corresponding to the eroded buffer condition (the cumulative distribution function listed in Table 4-1 for advective conditions), 2) commercial UNF fractional degradation rate (log-uniform distribution between $10^{-8}$ yr$^{-1}$ and $10^{-6}$ yr$^{-1}$), and 3) the far-field fracture zone length (uniform distribution between 500 and 1,700 m). The distribution of the three parameters were treated as epistemic uncertainty, and, for a given realization, a single value was sampled for each of the parameters. No correlation between sampled parameters was assumed.

As discussed in Section 4.1, erosion of the buffer is assumed to occur at 100,000 years, exposing 25 percent of the 3,000 waste packages in the capture zone to advective conditions. Only nine of the 1,000 realizations (0.9%) had sampled waste canister corrosion rates that were high enough to cause canister failures prior to one million years. Within each of the nine realizations, 750 waste packages (25 percent of the 3,000 waste packages in the capture zone) fail at the time calculated by the sampled corrosion rate.  Averaging over all realizations, the mean number of failed waste packages within one million years for this scenario is 6.75 (750 waste packages $\times$ 0.9%).  For simplification, all waste packages that fail are assumed to contain UNF.

Figure 4-11 shows the mean radionuclide mass release rate from the waste packages over time. The bumps in the curves correspond to each realization of waste package failure time (i.e., initiation of radionuclide release from 750 waste packages). Realization 216 has the earliest waste package failure time at about 209,000 years. Failure time for realization 888 occurs at about 306,000 years; that for realization 367 occurs at about 409,000 years, and so forth, with the

latest failure time occurring at 974,000 years for realization 127. The two radionuclides with the top mean release rates are [135]Cs and [129]I, which have unlimited solubilities and long half-lives ($2.3 \times 10^6$ yr and $1.7 \times 10^7$ yr, respectively). The radionuclides with the third and fourth highest mean release rates are [93]Nb and [107]Pd. Release rates generally increase with time as additional realizations with waste package failures are included with time. In general, radionuclides with long half-lives are released from the waste packages at appreciable rates; those with shorter half-lives decay markedly prior to waste package failure and are released at much lower rates. For example, for relatively short-lived ^{14}C, the peak of the mean waste package release rate is approximately $10^{-19}$ g yr$^{-1}$ at 210,000 years.
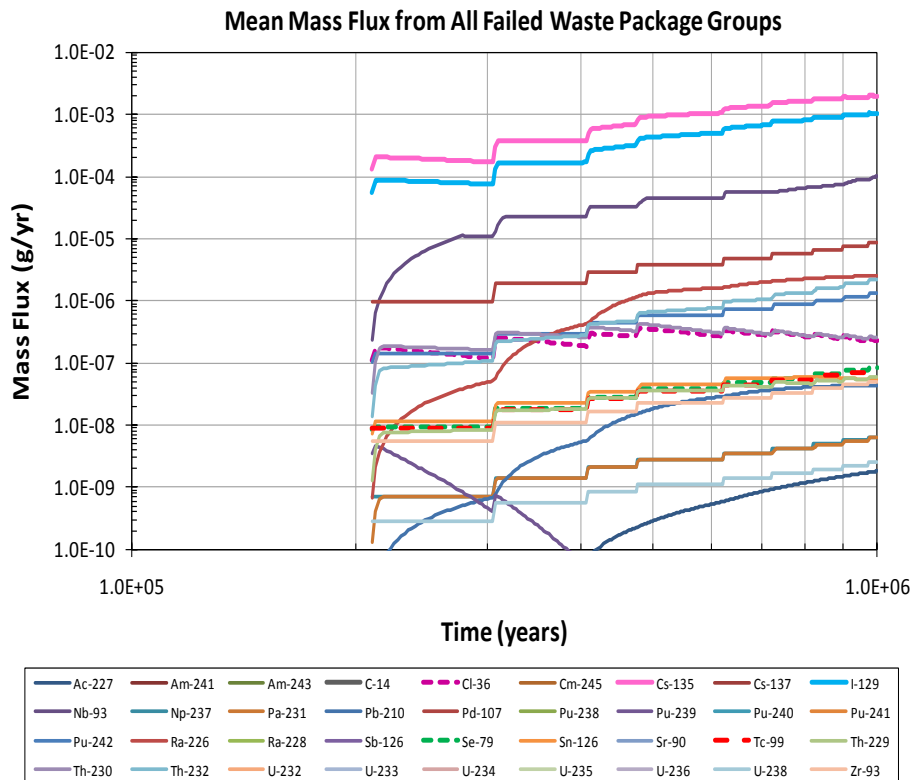


Figure 4-11.  Mean Radionuclide Mass Flux from Waste Packages in the Buffer Failure Scenario.

The mean mass fluxes released from the near-field fractures are shown in Figure 4-12. Only [129]I and [36]Cl are released at appreciable rates.  Other radionuclides that enter the near-field fractures at appreciable rates are amply retarded by adsorption. After an initial sharp increase to about $8 \times 10^{-5}$ g yr$^{-1}$ following the waste package failure of the realization with the earliest waste package failure time, the [129]I mean release rate generally increases for the rest of the analysis period to about $1.0 \times 10^{-3}$ g yr$^{-1}$ at one million years.  For [36]Cl, the mean release rate increases rapidly to a peak of $1.5 \times 10^{-7}$ g yr$^{-1}$ at about 232,000 years and remains relatively steady for the rest of the analysis period.
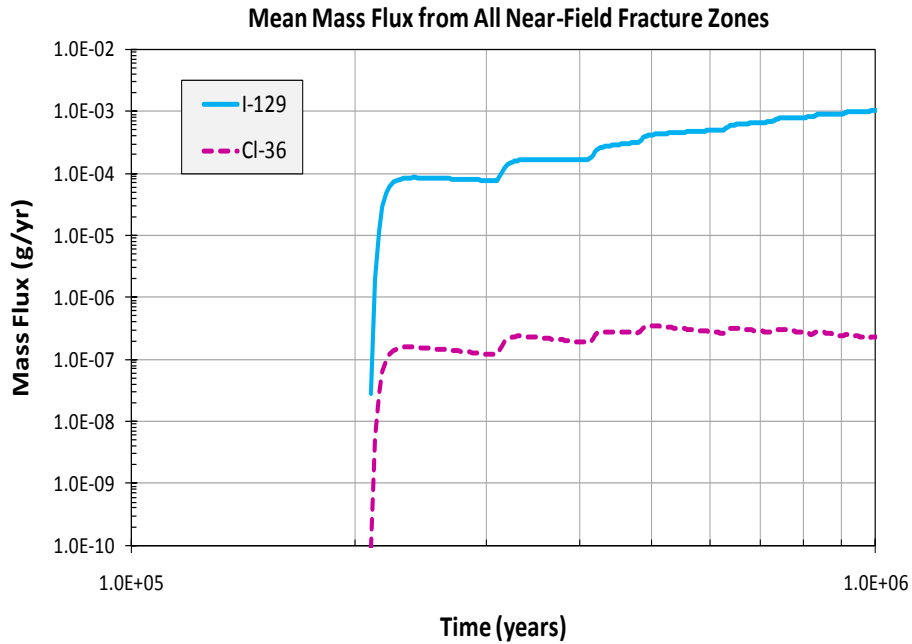
Figure 4-12. Mean Radionuclide Mass Flux from Near-Field Fractures in the Buffer Failure Scenario.

Mean releases of $^{129}$I and $^{36}$Cl from the far-field fracture zone (Figure 4-13) are similar to mean releases from the near-field fractures. The $^{129}$I mean release rate increases rapidly initially and then continues to increase for the rest of the analysis period. The peak mean mass flux of $^{129}$I is about $9.0 \times 10^{-4}$ g yr$^{-1}$ at one million years. As in the near-field granite release, following an initial rapid increase to a peak of about $1.5 \times 10^{-7}$ g yr$^{-1}$ at 440,000 years, the mean mass flux of $^{36}$Cl from the far-field fracture zone remains relatively steady, and the initial peak is delayed about 210,000 years compared to the release from the near-field granite.

**Mean Mass Flux from All far-Field Fracture Zones**
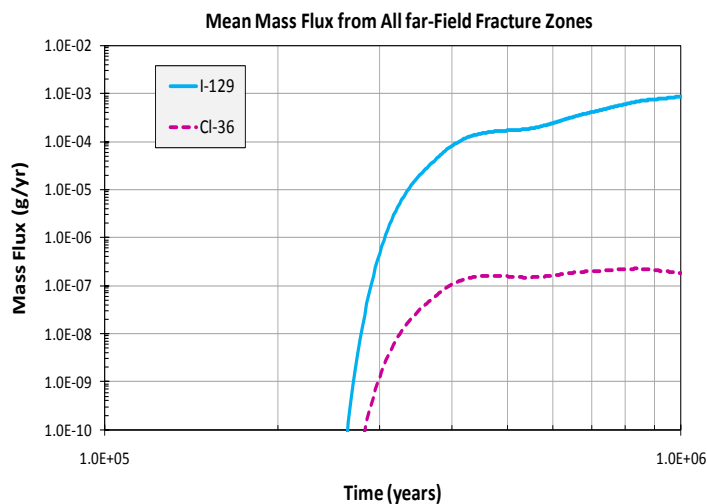
Figure 4-13.  Mean Radionuclide Mass Flux from Far-Field Fracture Conduits in the Buffer Failure Scenario.

Figure 4-14 shows the mean annual dose at the hypothetical accessible environment. $^{129}$I is the dominant dose contributor, and its contribution continues to increase over the entire analysis period. The peak dose rate from $^{129}$I is about 0.093 mrem yr$^{-1}$ at one million years. This dose rate is a factor of 3.4 higher than the dose rate from the defective waste package scenario at one million years (0.027 mrem yr$^{-1}$). The higher dose rate is mainly due to the higher mean number of failed waste packages at one million years (6.75). A greater relative dose rate difference that approximates the difference in the mean number of failed waste packages is not observed in the buffer failure scenario for two main reasons. First, the travel distance in the fracture zone increases from 500 m in the defective waste package scenario to a sampled value in the range of 500 m to 1,700 m. This increase causes the expected fracture zone residence time to more than double. Second, considering the longer expected residence time, a significant fraction of the waste package failures in the buffer failure scenario occur too late to have significant effects on dose at one million years. Also complicating the comparison is the switch from a fixed fractional waste form degradation rate to a sampled rate.
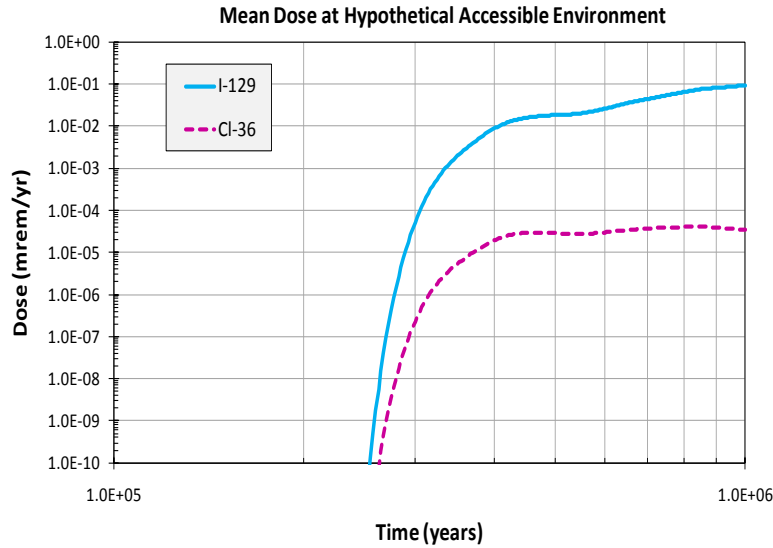
**Figure 4-14.** Mean Annual Radionuclide Dose at the Hypothetical Accessible Environment for the Buffer Failure Scenario.

## 4.3.  Relative Importance of Modeled FEPs

The conceptual model for the generic safety analysis presented in Section 4.1 can be considered a collection of FEPs that defines the release and transport of radionuclides from the repository to a logical human receptor. This collection may not include all potentially important FEPs and transport pathways for a given site nor may it by itself identify which of the included FEPs are of primary importance. However, by defining a simple conceptual model and numerically simulating it, we can evaluate which FEPs within the chosen set are most important.

The model developed for this report represents a single generalized conceptualization of a hypothetical repository, geosphere, and route of exposure. Because this model is not representative of the range of granite repository environments, EBS features, and exposure routes, a quantitative evaluation of importance was not performed. Instead, a qualitative evaluation was performed by visually comparing calculated radionuclide masses within the waste package, buffer, near field, and far field domains over time and releases from these domains over time.

The FEPs of primary and secondary importance, based on the qualitative evaluations, are summarized in Figure 4-15. The darker blue boxes indicate highly important features and processes, and the lighter boxes indicate features and processes of intermediate importance. The most important simulated processes for preventing release of radionuclides from the repository are canister corrosion, waste form degradation, and radionuclide precipitation. These processes, in turn, depend highly on reducing conditions and the presence and properties of the canister and buffer. The buffer, in addition to delaying waste package failure, presents a diffusive and sorptive barrier to radionuclide transport; however, the results suggest that the buffer's role in limiting canister corrosion rates is more important to the release of radionuclides to the

geosphere. Once the radionuclides enter the geosphere, fracture flow velocities, matrix diffusion, adsorption, and radioactive decay are of the highest importance to the dose rate at the receptor well.
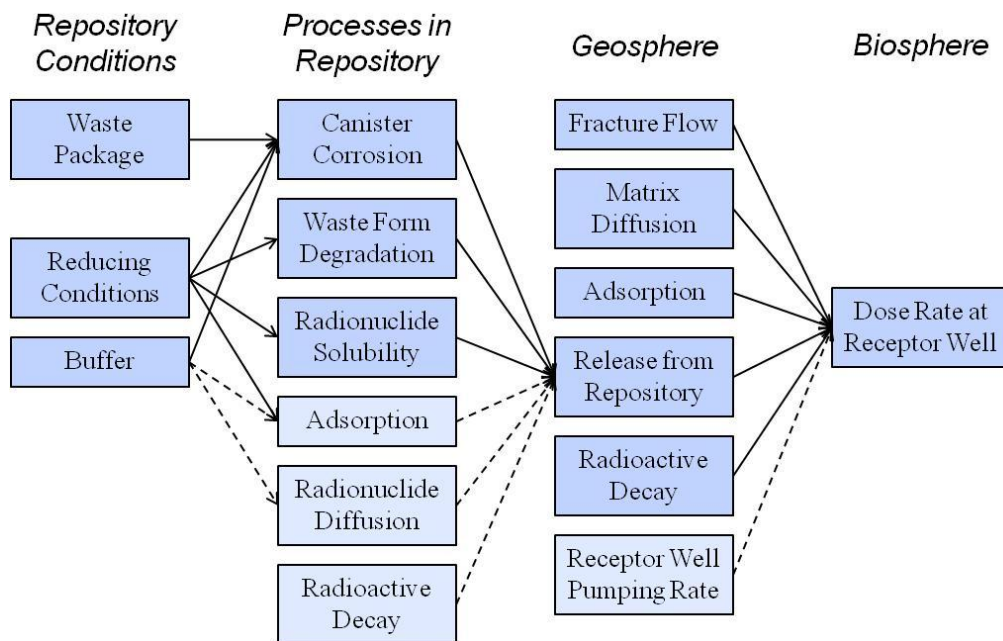


Figure 4-15.  Important Model Features and Processes.

The importance of radioactive decay is magnified by the long residence times in the repository and geosphere compared to the relatively short half-lives of many of the radionuclides. The long residence time in the waste package is due to slow canister corrosion rates, slow waste form degradation rates, and, for a few radioelements (e.g., uranium), solubility limits. Except in the defective waste package scenario, waste packages do not fail for more than 100,000 years. This delay in itself ensures that the initially high activities of $^{90}$Sr and $^{137}$Cs (Figure 2-1) decay to negligible quantities prior to waste package release. Because the waste form has a half-life of approximately seven million years (Section 4.2.1), the stability of the waste form contributes greatly to the repository residence time.

The model residence times of non-sorbing radionuclides in the near-field granite and far-field fracture zone are on the order of $10^4$ and $10^5$ years, respectively. These residence times are evident in Figure 4-7 and Figure 4-9 and are consistent with the pore velocities and path lengths of the modeled near-field and far-field conduits (Table 4-1 and Table 4-2). Because $^{14}$C and $^{36}$Cl have half-lives of 5,730 years and 301,000 years, respectively, significant fractions decay by the time the plumes of these radionuclides reach the well. In the buffer failure scenario, the additional residence time in the waste package is enough to prevent any appreciable breakthrough of $^{14}$C at the well (Figure 4-14). Little decay occurs within one million years for $^{129}$I, which has a half life of $1.7 \times 10^7$ years. Thus, the dose rate over time at the receptor well for both scenarios is essentially entirely due to $^{129}$I.

While processes and rates in the EBS and geosphere are uncertain in many respects, they are arguably not as uncertain as the future timing and placement of potential receptors in the

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biosphere. Future human intrusion, placement of future wells, and the size and proximity of future human settlements near the repository are highly uncertain. Nevertheless, receptors have to be defined in order to estimate dose rates over time. For the Yucca Mountain repository, 40 CFR 197 specifies a RMEI living along the predominant groundwater flow path 18 km from the Yucca Mountain site. For the current analysis, the receptor is located at a distance of 500 m.

The choice of 500 m is somewhat arbitrary because an actual site has not been identified and specific regulations for a repository in granite have not been developed. Locating the receptor within 500 m of the repository is conservative in that it reduces the residence time in the geosphere; however, it is not the only choice that must be made regarding the receptor. Other choices include well discharge rate, use of well water, receptor diet and lifestyle, and fraction of the repository within the capture zone of the well.

For the scenarios simulated here, the receptor well is assumed to serve a single self-sufficient farming family and pump at a rate of 738 m$^3$ yr$^{-1}$. The well is assumed to capture flow from 3,000 emplacement boreholes spread out over approximately 0.7 km$^2$. These assumptions are largely consistent with the flow model developed by Gierzewski et al. (2004). Most of the pumpage is expected to originate from shallower depths where the water is fresh. The total water flux into the well originating from the fracture zone conduits down-gradient of the 3,000 emplacement boreholes is simulated to be 0.3 m$^3$ yr$^{-1}$, which is 0.04% of the well discharge rate. This low percentage is reasonable if the vast majority of the discharge is from fresh water sources, such as shallow recharge. In this case, the fresh water would dilute the saline water from the repository zone (and its radionuclide content) by a factor of 2,460.

The dilution of the captured fracture zone water at the well does not include the dilution caused by the mixing of water in the fracture zone down-gradient of the 3,000 emplacement boreholes. In the model, radionuclides released from a waste package emanate from a single fracture intersecting the emplacement borehole. The flow rate for this fracture is 0.0001 m$^3$ yr$^{-1}$. Thus, neglecting dispersion and the additional dilution that occurs in the emplacement borehole, the overall dilution factor at the well for radionuclides released from a single waste package is 7,380,000.

In the unlikely case that the well captures flow from 30,000 emplacement boreholes (i.e., nearly the entire UNF inventory), the well would need to pump at a much greater discharge rate. Increasing the discharge rate to capture flow from 30,000 emplacement boreholes would increase the radionuclide mass flux by a similar factor. However, the concentrations at the well would be less affected because the greater radionuclide mass flux would be largely offset by the greater discharge rate. This relationship between capture zone size and well discharge rate implies that the dilution factor and dose rate are not highly sensitive to capture zone size. An additional factor that would contribute to low dose rates over time for a larger capture zone would be the expanded range of radionuclide travel distances.

In addition to the important features and processes of the model shown in Figure 4-15, there are several important simplifying assumptions. When assumptions were made, the goal was to make them reasonable and realistic. However, conservative assumptions were adopted in some instances for simplicity and to accommodate uncertainty in a generic assessment. The more conservative assumptions in the analyses include:

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- *Complete disappearance of the copper canister, insert, and fuel cladding once the copper canister is breached*. This assumption is especially conservative in the defective waste package scenario because it implies immediate release of radionuclides even though corrosion of the insert and fuel cladding would delay releases. For example, data indicate that Zircaloy cladding may have a lifetime of at least 100,000 years (SKB 2006c, Section 2.5.4).

- *Corrosion products from the canister and its contents do not retard radionuclide release*. The release of many radionuclides from the waste package is likely to be retarded due to strong adsorption to corrosion products.

- *No lateral dispersion*. All radionuclides remain within the confines of the modeled conduits and migrate toward the receptor well.

- *In the buffer failure scenario, complete removal of the buffer for one fourth of the waste packages after the first glacial period*. This fraction was based on the probability distributions presented by SKB (2006d, Figure 9-99) assuming spalling. The safety assessment in SKB's license application submitted in March 2011 estimates a much lower fraction in the reference evolution consistent with limited or no spalling. Within one million years, the SKB PA estimates that approximately 0.4% of deposition boreholes become exposed to advective conditions in the reference evolution. In the most unfavorable cases simulated, this percentage increases but remains below 10% (SKB 2011, Section 12.2).

These assumptions increase the likelihood and magnitude of calculated dose rates, either by accelerating the release of radionuclides, accelerating the migration of radionuclides toward the receptor well, or limiting attenuation of radionuclides along the flow paths. The model would be improved by incorporating realistic representations and associated uncertainty in place of these assumptions because doing so would provide a more realistic sense of overall performance. However, despite the conservative assumptions, dose rates projected in these scenarios are well within acceptable limits. The calculated dose rates shown in Figure 4-10 and Figure 4-14 are less than 0.2% of current regulatory limits (Section 1.2).

## 4.4. Comparison of Results to Other Safety Assessments

Generic safety assessments have been performed for other deep disposal concepts including deep borehole, mined salt, and mined clay/shale (Hansen et al. 2010; Brady et al. 2009; Hansen and Leigh 2011). In addition, in-depth site-specific assessments for repositories in granite have been performed (Gierszewski et al. 2004; SKB 2011; Posiva 2008b). These studies are compared with the current study to assess differences and similarities of model concepts and calculations.

A primary difference between the safety assessment presented here and in generic assessments for deep borehole and non-granite repositories is the role of the waste package and waste form. In the deep borehole concept, waste packages are emplaced in granite bedrock at depths of 3 to 5 km. Because the permeability at this depth is extremely low ($\sim 10^{-20}$ m$^2$), the generic model does not include the performance of the waste package or the waste form (Brady et al. 2009). Despite these simplifications, the calculated dose rates from a 1 km deep well located above the disposal

zone remains approximately 10 orders of magnitude below current criteria.  As in the results of the granite model in this study (Figure 4-10 and Figure 4-14), [129]I is the only significant contributor to the dose rate.

The generic clay/shale disposal model of Hansen et al. (2010) also assumes immediate degradation and exposure of the waste form and relies heavily on the low permeability ($\sim 10^{-19}$ m$^2$) of the host rock. An inventory of 200,000 waste packages is assumed to be emplaced 150 m deep in the clay/shale host rock, 450 m below land surface. Radionuclides slowly diffuse through the host rock into an overlying aquifer where they are captured by a domestic well pumping at a rate of 3,000 acre-feet yr$^{-1}$. The timing and magnitude of the peak annual dose are similar to those calculated in the defective waste package scenario of the current study (Figure 4-10). In both calculations, the dose rate within this time frame is essentially entirely due to [129]I. However, unlike the results of the granite model, the annual dose declines rapidly at approximately 300,000 years, whereupon other radionuclides become major contributors. Much of this difference is likely due to the assumption of instantaneous waste form degradation in the clay/shale model. In the granite model, the waste form has a half-life of approximately seven million years, so radionuclides continue to be released over time.

Salt as a repository host rock has advantages similar in several ways to clay/shale. Its permeability is extremely low and would strongly inhibit radionuclide transport. Low permeability in these formations is enhanced by deformation properties that promote closure of large voids and fracture healing. Hansen and Leigh (2011) present a detailed technical basis for a salt repository. Their report does not offer a quantitative generic safety assessment. Instead, it identifies specific areas where further research and model development are needed to inform quantitative simulations.

The results of the defective canister scenario of the granite repository safety assessment performed by Gierszewski et al. (2004, Figure 7.7) are quite similar to those of the current study (Figure 4-10). This is not surprising because the pore velocities, travel distances, well discharge rate, and approximate repository capture zone of the well in the generic model are based on the results of their reference flow model. The peak dose rates for their reference case and high permeability geosphere case are $\sim$0.01 mrem yr$^{-1}$ at $\sim$400,000 years and $\sim$0.03 mrem yr$^{-1}$ at $\sim$100,000 years, respectively. There is also good agreement on the radionuclide contributions to dose: [129]I is the primary contributor and [36]Cl is a distant second.

Groundwater residence time in granite host rock has a broad range in PAs. The FRAC3DVS flow model of Gierszewski et al. (2004) assumed a permeability of $7 \times 10^{-19}$ m$^2$ at repository depth in the reference case and resulted in a groundwater velocity at that depth of approximately 0.001 m yr$^{-1}$. Much higher pore velocities and much shorter residence times in the geosphere are simulated in the safety assessment of the SKB license application for a granite repository in Forsmark. In the central corrosion case of that assessment, [226]Ra, a radionuclide that adsorbs appreciably but not strongly to buffer and granite, is a primary contributor to dose at the receptor well (SKB 2011, Section 13.5). [226]Ra has a half-life of 1,600 years and can only be important to dose if it reaches the well within several half-lives. In-growth of [226]Ra cannot explain its high activity at the receptor because its parent, [230]Th, is much less mobile in the geosphere. Thus, water travel time from the repository to the biosphere in the SKB model appears to be

approximately two orders of magnitude (or more) lower than the generic granite model of the current report.

The other radionuclides with significant contributions to dose in the central corrosion case of the SKB license application include $^{129}$I, $^{237}$Np, $^{79}$Se, and $^{59}$Ni. The first three originate from the waste form. Neptunium and selenium mildly adsorb along the flow path, but their low retardation combined with the relatively short water travel time allows them to reach the biosphere in less than 100,000 years. $^{59}$Ni is produced by the neutron activation of stable $^{58}$Ni in various metal parts in the waste package (SKB 2006b, Section 3.2). Because the entire inventory of $^{59}$Ni is assumed to be released upon canister failure, spikes of $^{59}$Ni are observed in the far field soon after canisters fail (SKB 2011, Section 13.5). The release of radionuclides from metal parts is not simulated in the generic safety assessment of the current report.

The far-field mean annual effective dose as a function of time in the central corrosion case of the SKB analysis is similar to that of $^{129}$I in Figure 4-14. The dose rate increases rapidly after initial breakthrough between 100,000 and 200,000 years and then more slowly as the number of failed canisters increases with time.  At one million years, the mean dose rate for the SKB case is 0.02 mrem yr$^{-1}$ (SKB 2011, Figure 13-19), one fourth of the rate in Figure 4-14 at the same point in time.

A much lower peak dose rate is calculated for the defective canister scenario of a PA for a repository at Olkiluoto, Finland (Posiva 2008b, Section 7.2). In that assessment, a 1-mm hole in the canister leads to a total dose rate that peaks near $10^{-5}$ mrem yr$^{-1}$ at approximately 7,000 years and then again at one million years. $^{129}$I is the primary contributor to the annual dose except for a short time period around the 7,000-year peak when the dose rate from $^{14}$C reaches its maximum. The early breakthrough is a result of combining early waste package failure with relatively high groundwater velocities.

Overall, the effective dose rate comparisons show strong agreement and build confidence in the ability of the generic granite model to simulate major FEPs related to deep disposal of radioactive waste in granite. With a working model, uncertainty and sensitivity analyses can be performed to evaluate the relative importance of the included FEPs and to identify parameters that propagate significant uncertainty to model results. Such analyses are not included in the current report.

# 5. SUMMARY AND CONCLUSIONS

Granite is common across the U.S. at depths suitable for a mined repository for nuclear waste. A repository in granite that incorporates the KBS-3 concept is anticipated to satisfy performance objectives, including retrievability if deemed important. Chemically reducing conditions, buffer-limited advection of groundwater, and long-term stability of excavations would provide a physical and chemical environment that would likely preserve copper waste canisters for millions of years. Because of this ability of a granite repository to preserve waste canisters, canister failure in less than one million years would have a low probability, affecting a small fraction of the waste packages. For those that fail, slow waste form degradation, low solubility, and adsorption to buffer materials would substantially delay release of much of the radionuclide inventory from the repository and allow for significant radioactive decay. Beyond the repository, the low permeability of the granite host rock would strongly inhibit radionuclide transport to the biosphere.

## 5.1.  Summary of Findings

Boreholes and mined excavations in granite within the U.S. and around the world indicate that granite often exists with few fractures and low permeability at depths below 300 m. Because large bodies of granite are found across the U.S., there are likely numerous granite sites in the U.S. with favorable characteristics for a mined repository.

A repository in granite has a strong technical basis. Unlike disposal concepts for salt, shale/clay, or deep boreholes, the technical basis for granite is rooted in the ability of the granite repository to preserve waste packages. Preservation of waste packages depends on the mechanical stability of granite excavations, performance of the buffer and canister, and favorable chemical conditions at repository depths. The KBS-3 design, proposed for the Swedish repository and included in Finnish and Canadian concepts, calls for waste to be encased in copper canisters and for a densely-packed clay buffer to surround the canisters. The diffusive barrier presented by the buffer is anticipated to prevent corrosive reactants, such as hydrogen sulfide, from reaching the canisters at rates needed to cause canister failure within one million years. The buffer is anticipated to be preserved by low groundwater fluxes, favorable chemical conditions, backfill, and the rigid confines of the host rock. An added potential advantage of disposal in mined granite is that waste packages would be retrievable far into the future, should such an objective be specified.

The results of the generic safety assessment developed in this report indicate that a granite repository would satisfy established safety criteria. They also suggest that a small number of FEPs would largely control the release and transport of radionuclides. Canister degradation, waste form degradation, chemical environment, buffer performance, radionuclide solubility, radionuclide adsorption, and radionuclide decay are all highly important to radionuclide release from a granite repository. In the geosphere, the important features and processes include fracture flow rates, matrix diffusion, and adsorption. The results imply that these FEPs should be well understood when selecting a site for a repository and developing a site-specific PA model.

A proposed site for a granite repository would require a specific design concept, reliable data, and a detailed PA. Important hydrogeologic data would include spatially distributed

permeability, flow porosity, diffusion porosity, groundwater composition, and the locations of major fractures. These data would be used in a regional hydrologic model to predict groundwater velocities and travel times for current and potential future climates. Important repository data would include waste package inventories, canister corrosion rates, waste form degradation rates, and buffer properties (e.g., permeability, adsorption of radionuclides, and properties related to resilience when subjected to potential environmental conditions). These and other data related to processes identified by FEP screening would be incorporated in a total system PA model to determine whether overall performance criteria can be met.

Safety assessments have been performed for specific sites in Sweden, Finland, and Canada (SKB 2011; Posiva 2008b; Gierszewski et al. 2004). The Swedish safety assessment supports the world's first license application for a granite repository, submitted in March of 2011(SKB 2011). A license application for a granite repository in Finland is anticipated in 2012. Each of these safety assessments conclude that a granite repository using the KBS-3 or similar design concept would perform adequately. These conclusions are consistent with the preliminary conclusions of the generic safety analysis presented in this report.

Due to the favorable results from safety assessment models as well as the widespread availability of suitable granite bodies across the U.S., disposal of HLW in a mined granite repository is a feasible option for the U.S. Much experimental work has been performed, mostly by international programs and commonly in underground laboratories built in crystalline rock. This research is extensive and contributes greatly to our understanding of important FEPs in the repository and granite host rock. These studies are publicly available and would provide a valuable resource should a repository in granite be reconsidered in the U.S.

## 5.2. Recommendations

Calibration and validation of models used in the design, operation, and performance of an underground repository require data obtained from underground studies. In the 1980s, experimental work at the underground research facility constructed in the quartz monzonite at the Nevada Test Site produced valuable data related to the performance of crystalline rock as a host rock for a HLW repository (e.g., Patrick 1986). Should the U.S. decide to pursue granite as a potential repository host rock, an underground research laboratory located in the U.S. would be highly beneficial. In addition, U.S. repository programs could learn much from the operations and experimental work conducted at existing underground laboratories in other countries and could pursue additional research at these facilities.

The selection of a site (or sites) for an underground laboratory in the U.S. should place a high priority on locations that are both politically and technically viable. To help identify repository options for willing communities and states, geographic data across the U.S. should be collected on key characteristics of crystalline rock at disposal depth. Key characteristics include permeability, diffusion porosity, fracturing and fracture zone characteristics, vertical and lateral extent of the granite, mineral composition, mechanical and thermal properties, groundwater composition, groundwater age, redox conditions, and in situ stress.

Engineered barriers have important roles in the total system performance of a mined granite repository. Therefore, the long-term performance of the waste canister, fuel cladding, waste

form, and buffer under potential repository conditions should be well understood. Continued research in the degradation of these barriers and the range of possible repository conditions is needed to ensure accurate simulation of engineered barrier performance. In addition, advanced engineered designs, such as the prefabrication design suggested by McKinley et al. (2000), should be evaluated for potential improvement to the KBS-3 design.

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# APPENDIX A:  U.S. NUCLEAR WASTE INVENTORY

Three different types of HLW were considered in the source-term model: commercial UNF, existing DOE HLW, and reprocessing HLW of the commercial UNF. The radionuclide inventory presented in this appendix is based on the detailed fuel cycle waste inventory analysis data presented in Carter and Luptak (2010). The inventory consists of actinide elements in several radionuclide decay chains along with a number of fission products.

## A.1   Commercial Used Nuclear Fuel Inventory

The once-through fuel cycle waste inventory analysis considered four scenarios to evaluate the projected increases in the commercial light water reactor (LWR) UNF inventory. The scenarios were considered to provide a wide range of LWR fuel inventory for use in future analysis (Carter and Luptak 2010, Section 3.2). The inventory analysis used Scenario 1, which assumes no replacement of existing nuclear generation reactors. Selection of this particular scenario for the inventory analysis is arbitrary, and it may be revised in future analysis. For this scenario, a total of 140,000 metric tons uranium (MTU) used fuel is estimated to be discharged from reactors by the year 2055 (Carter and Luptak 2010, Table 3-5). Out of the total inventory, 91,000 MTU is for the pressurized water reactors (PWRs) used fuels with an estimated total of 209,000 assemblies. This is equivalent to 0.435 MTU per PWR assembly.

For simplification of the source-term inventory analysis, the total inventory was converted to the equivalent PWR inventory, resulting in a total of 321,540 PWR assemblies. The source-term model assumes that a waste package (WP) contains 10 PWR assemblies, and a total of 32,154 waste packages are needed to dispose of the commercial UNF.

The isotope inventory of the UNF is assumed to be represented by the PWR fuel with a burn-up of 60 GWd/MTIHM and 4.73% enrichment and aged 30 years after discharge from reactor (Carter and Luptak 2010, Table C-1). The isotope inventory for the radionuclides of the commercial UNF included in the source-term model is shown in Table A-1.

## A.2   DOE High-Level Radioactive Waste

All existing DOE HLW is assumed to be immobilized in the borosilicate glass logs. The source-term inventory analysis used the best-estimate projected total number of DOE HLW canisters documented in the fuel cycle inventory analysis report (Carter and Luptak 2010, Table 2-2); the best estimate projection is 25,016 canisters. The source-term model assumes that each waste package contains 5 HLW canisters, and a total of 5,003 waste packages are needed to dispose of the DOE HLW.

The isotope inventory of the DOE HLW is given for each radionuclide in terms of the total radioactivity ($Ci$) in the fuel cycle inventory analysis report (Carter and Luptak 2010, Table F-1). The radioactivity was converted to the equivalent mass ($m_i$) for each radionuclide as follows:

$$m_i \ (g) = \frac{A_i \cdot t_{1/2,i} \cdot MW_i}{0.693 \cdot N_A} \ , \tag{A-1}$$

where $A_i$ is the radioactivity of radionuclide $i$, $t_{1/2,i}$ is the half-life of radionuclide $i$, $MW_i$ is the molecular weight of radionuclide $i$, and $N_A$ is the Avogadro constant ($6.023 \times 10^{23}$). The total mass of radionuclides of the existing DOE HLW is estimated to be 1,759 MT. This results in 0.07 MT of radionuclides per HLW canister, and 0.35 MT of radionuclides per waste package. The isotope inventory per HLW canister and per waste package for the radionuclides included in the source-term model is given in Table A-2.

## A.3   Reprocessing High-Level Radioactive Waste

The fuel cycle inventory analysis report discusses several candidate reprocessing methods for commercial UNF and their potential waste streams (Carter and Luptak 2010, Section 4). For simplification of the source-term model analysis, the following assumptions were made for "hypothetical" reprocessing of commercial UNF:

- Ninety nine percent (99%) of uranium and plutonium are recovered. All others including transuranic elements and fission products of the commercial UNF inventory (140,000 MTU) remain in the waste streams.

- Reprocessing HLW is immobilized in borosilicate glass as for the DOE HLW.

- Reprocessing HLW is encapsulated at the same radionuclide mass loading as for the DOE HLW (i.e., 0.07 MT radionuclide mass per canister).

Note that the above assumptions result in higher concentrations of fission products in the hypothetical reprocessing waste streams and glass waste form than the DOE HLW.
The total radionuclide mass of the hypothetical reprocessing HLW is estimated to be 1,426 MT (after removing 99% of uranium and plutonium). With a radionuclide mass loading of 0.07 MT per canister, this is equivalent to a total of 20,276 canisters. The source-term model assumes that each waste package contains five reprocessing HLW canisters, and a total of 4,055 waste packages are needed for disposal. The isotope inventory per reprocessing HLW canister and per waste package for the radionuclides included in the source-term model is given in Table A-3.

## A.4   References

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Table A-1.  Isotope Inventory for Commercial UNF Used for Source-Term Model Analysis.

| Isotope | Half Life (years) | Fractional Mass Inventory | Isotope mass per WP (g) |
|---------|-------------------|---------------------------|--------------------------|
| Ac-227 | 2.18E+01 | 2.7469E-13 | 1.1960E-06 |
| Am-241 | 4.32E+02 | 8.7003E-04 | 3.7882E+03 |
| Am-243 | 7.37E+03 | 1.8796E-04 | 8.1841E+02 |
| C-14 | 5.71E+03 | 3.1524E-07 | 1.3726E+00 |
| Cl-36 | 3.01E+05 | 3.4808E-07 | 1.5156E+00 |
| Cm-245 | 8.50E+03 | 6.6221E-06 | 2.8833E+01 |
| Cs-135 | 2.30E+06 | 5.3570E-04 | 2.3325E+03 |
| Cs-137 | 3.01E+01 | 7.2561E-04 | 3.1593E+03 |
| I-129 | 1.70E+07 | 2.1754E-04 | 9.4720E+02 |
| Nb-93 | 1.36E+01 | 4.9591E-04 | 2.1592E+03 |
| Np-237 | 2.14E+06 | 8.5892E-04 | 3.7398E+03 |
| Pa-231 | 3.25E+04 | 7.1103E-10 | 3.0959E-03 |
| Pb-210 | 2.26E+01 | 7.8324E-15 | 3.4103E-08 |
| Pd-107 | 6.50E+06 | 2.8663E-04 | 1.2480E+03 |
| Pu-238 | 8.77E+01 | 3.4170E-04 | 1.4878E+03 |
| Pu-239 | 2.41E+04 | 5.1487E-03 | 2.2418E+04 |
| Pu-240 | 6.54E+03 | 2.8427E-03 | 1.2377E+04 |
| Pu-241 | 1.44E+01 | 2.6198E-04 | 1.1407E+03 |
| Pu-242 | 3.76E+05 | 5.6750E-04 | 2.4709E+03 |
| Ra-226 | 1.60E+03 | 2.2081E-12 | 9.6141E-06 |
| Ra-228 | 6.70E+00 | 1.4339E-18 | 6.2431E-12 |
| Sb-126 | 3.61E-05 | 1.6470E-12 | 7.1713E-06 |
| Se-79 | 2.95E+05 | 7.2769E-06 | 3.1684E+01 |
| Sn-126 | 1.00E+05 | 3.4663E-05 | 1.5092E+02 |
| Sr-90 | 2.91E+01 | 3.0809E-04 | 1.3414E+03 |
| Tc-99 | 2.13E+05 | 8.8739E-04 | 3.8638E+03 |
| Th-229 | 7.90E+03 | 4.4252E-12 | 1.9267E-05 |
| Th-230 | 7.54E+03 | 1.5838E-08 | 6.8961E-02 |
| Th-232 | 1.41E+10 | 4.2412E-09 | 1.8466E-02 |
| U-232 | 6.89E+01 | 3.1642E-09 | 1.3777E-02 |
| U-233 | 1.59E+05 | 9.7002E-09 | 4.2235E-02 |

| Isotope | Half Life (years) | Fractional Mass Inventory | Isotope mass per WP (g) |
|---------|-------------------|---------------------------|-------------------------|
| U-234 | 2.45E+05 | 2.1220E-04 | 9.2392E+02 |
| U-235 | 7.04E+08 | 3.7329E-03 | 1.6253E+04 |
| U-236 | 2.34E+07 | 4.3349E-03 | 1.8874E+04 |
| U-238 | 4.46E+09 | 6.3215E-01 | 2.7524E+06 |
| Zr-93 | 1.53E+06 | 1.0193E-03 | 4.4382E+03 |

Table A-2.  Isotope Inventory for DOE HLW Used for Source-Term Model Analysis.

| Isotope | Half Life (years) | Fractional Mass Inventory | Isotope mass per canister (g) | Isotope mass per WP (g) |
|---------|-------------------|---------------------------|-------------------------------|-------------------------|
| Ac-227 | 2.18E+01 | 1.139E-09 | 8.010E-05 | 4.005E-04 |
| Am-241 | 4.32E+02 | 4.022E-04 | 2.829E+01 | 1.414E+02 |
| Am-243 | 7.37E+03 | 2.732E-05 | 1.922E+00 | 9.608E+00 |
| C-14 | 5.71E+03 | 1.747E-08 | 1.228E-03 | 6.142E-03 |
| Cl-36 | 3.01E+05 | 0.000E+00 | 0.000E+00 | 0.000E+00 |
| Cm-245 | 8.50E+03 | 5.428E-07 | 3.817E-02 | 1.909E-01 |
| Cs-135 | 2.30E+06 | 1.759E-03 | 1.237E+02 | 6.184E+02 |
| Cs-137 | 3.01E+01 | 2.219E-03 | 1.561E+02 | 7.804E+02 |
| I-129 | 1.70E+07 | 1.802E-04 | 1.268E+01 | 6.338E+01 |
| Nb-93 | 1.36E+01 | 0.000E+00 | 0.000E+00 | 0.000E+00 |
| Np-237 | 2.14E+06 | 3.004E-04 | 2.113E+01 | 1.056E+02 |
| Pa-231 | 3.25E+04 | 3.452E-06 | 2.427E-01 | 1.214E+00 |
| Pb-210 | 2.26E+01 | 1.317E-13 | 9.264E-09 | 4.632E-08 |
| Pd-107 | 6.50E+06 | 2.188E-05 | 1.539E+00 | 7.696E+00 |
| Pu-238 | 8.77E+01 | 2.070E-04 | 1.456E+01 | 7.279E+01 |
| Pu-239 | 2.41E+04 | 1.749E-03 | 1.230E+02 | 6.151E+02 |
| Pu-240 | 6.54E+03 | 1.865E-04 | 1.312E+01 | 6.559E+01 |
| Pu-241 | 1.44E+01 | 2.468E-06 | 1.736E-01 | 8.678E-01 |
| Pu-242 | 3.76E+05 | 2.154E-05 | 1.515E+00 | 7.573E+00 |
| Ra-226 | 1.60E+03 | 5.747E-11 | 4.042E-06 | 2.021E-05 |
| Ra-228 | 6.70E+00 | 4.563E-11 | 3.209E-06 | 1.604E-05 |
| Sb-126 | 3.61E-05 | 5.728E-12 | 4.029E-07 | 2.014E-06 |
| Se-79 | 2.95E+05 | 3.085E-04 | 2.169E+01 | 1.085E+02 |
| Sn-126 | 1.00E+05 | 1.215E-04 | 8.548E+00 | 4.274E+01 |

| Isotope | Half Life (years) | Fractional Mass Inventory | Isotope mass per canister (g) | Isotope mass per WP (g) |
|---------|-------------------|---------------------------|-------------------------------|--------------------------|
| Sr-90 | 2.91E+01 | 9.262E-04 | 6.514E+01 | 3.257E+02 |
| Tc-99 | 2.13E+05 | 3.212E-03 | 2.259E+02 | 1.129E+03 |
| Th-229 | 7.90E+03 | 9.980E-09 | 7.019E-04 | 3.509E-03 |
| Th-230 | 7.54E+03 | 4.546E-09 | 3.197E-04 | 1.599E-03 |
| Th-232 | 1.41E+10 | 9.894E-02 | 6.958E+03 | 3.479E+04 |
| U-232 | 6.89E+01 | 1.141E-09 | 8.022E-05 | 4.011E-04 |
| U-233 | 1.59E+05 | 5.300E-05 | 3.727E+00 | 1.864E+01 |
| U-234 | 2.45E+05 | 7.431E-05 | 5.226E+00 | 2.613E+01 |
| U-235 | 7.04E+08 | 3.732E-03 | 2.625E+02 | 1.312E+03 |
| U-236 | 2.34E+07 | 2.863E-04 | 2.014E+01 | 1.007E+02 |
| U-238 | 4.46E+09 | 8.821E-01 | 6.204E+04 | 3.102E+05 |
| Zr-93 | 1.53E+06 | 1.739E-03 | 1.223E+02 | 6.115E+02 |

Table A-3.  Isotope Inventory for Reprocessing HLW Used for Source-Term Model Analysis.

| Isotope | Half Life (years) | Fractional Mass Inventory | Isotope mass per canister (g) | Isotope mass per WP (g) |
|---------|-------------------|---------------------------|-------------------------------|--------------------------|
| Ac-227 | 2.18E+01 | 2.6969E-11 | 1.8967E-06 | 9.4833E-06 |
| Am-241 | 4.32E+02 | 8.5419E-02 | 6.0073E+03 | 3.0037E+04 |
| Am-243 | 7.37E+03 | 1.8454E-02 | 1.2978E+03 | 6.4892E+03 |
| C-14 | 5.71E+03 | 3.0950E-05 | 2.1766E+00 | 1.0883E+01 |
| Cl-36 | 3.01E+05 | 0.0000E+00 | 0.0000E+00 | 0.0000E+00 |
| Cm-245 | 8.50E+03 | 6.5015E-04 | 4.5724E+01 | 2.2862E+02 |
| Cs-135 | 2.30E+06 | 5.2594E-02 | 3.6989E+03 | 1.8494E+04 |
| Cs-137 | 3.01E+01 | 7.1239E-02 | 5.0101E+03 | 2.5051E+04 |
| I-129 | 1.70E+07 | 2.1358E-02 | 1.5021E+03 | 7.5104E+03 |
| Nb-93 | 1.36E+01 | 6.8717E-07 | 4.8327E-02 | 2.4164E-01 |
| Np-237 | 2.14E+06 | 8.4328E-02 | 5.9306E+03 | 2.9653E+04 |
| Pa-231 | 3.25E+04 | 6.9808E-08 | 4.9094E-03 | 2.4547E-02 |
| Pb-210 | 2.26E+01 | 7.6897E-13 | 5.4080E-08 | 2.7040E-07 |
| Pd-107 | 6.50E+06 | 2.8141E-02 | 1.9791E+03 | 9.8956E+03 |
| Pu-238 | 8.77E+01 | 3.3547E-05 | 2.3593E+00 | 1.1797E+01 |
| Pu-239 | 2.41E+04 | 5.0549E-04 | 3.5550E+01 | 1.7775E+02 |
| Pu-240 | 6.54E+03 | 2.7909E-04 | 1.9628E+01 | 9.8141E+01 |

| Isotope | Half Life (years) | Fractional Mass Inventory | Isotope mass per canister (g) | Isotope mass per WP (g) |
|---|---|---|---|---|
| Pu-241 | 1.44E+01 | 2.5721E-05 | 1.8089E+00 | 9.0446E+00 |
| Pu-242 | 3.76E+05 | 5.5717E-05 | 3.9184E+00 | 1.9592E+01 |
| Ra-226 | 1.60E+03 | 2.1679E-10 | 1.5246E-05 | 7.6230E-05 |
| Ra-228 | 6.70E+00 | 1.4077E-16 | 9.9004E-12 | 4.9502E-11 |
| Sb-126 | 3.61E-05 | 1.6170E-10 | 1.1372E-05 | 5.6861E-05 |
| Se-79 | 2.95E+05 | 7.1444E-04 | 5.0245E+01 | 2.5122E+02 |
| Sn-126 | 1.00E+05 | 3.4031E-03 | 2.3933E+02 | 1.1967E+03 |
| Sr-90 | 2.91E+01 | 3.0248E-02 | 2.1273E+03 | 1.0636E+04 |
| Tc-99 | 2.13E+05 | 8.7123E-02 | 6.1272E+03 | 3.0636E+04 |
| Th-229 | 7.90E+03 | 4.3446E-10 | 3.0554E-05 | 1.5277E-04 |
| Th-230 | 7.54E+03 | 1.5550E-06 | 1.0936E-01 | 5.4680E-01 |
| Th-232 | 1.41E+10 | 4.1639E-07 | 2.9284E-02 | 1.4642E-01 |
| U-232 | 6.89E+01 | 3.1066E-10 | 2.1848E-05 | 1.0924E-04 |
| U-233 | 1.59E+05 | 9.5236E-10 | 6.6977E-05 | 3.3489E-04 |
| U-234 | 2.45E+05 | 2.0833E-05 | 1.4652E+00 | 7.3258E+00 |
| U-235 | 7.04E+08 | 3.6649E-04 | 2.5775E+01 | 1.2887E+02 |
| U-236 | 2.34E+07 | 4.2559E-04 | 2.9931E+01 | 1.4966E+02 |
| U-238 | 4.46E+09 | 6.2063E-02 | 4.3648E+03 | 2.1824E+04 |
| Zr-93 | 1.53E+06 | 1.0008E-01 | 7.0381E+03 | 3.5191E+04 |

# APPENDIX B: FEATURES, EVENTS, AND PROCESSES

## B.1 FEPs and Likely Screening Decisions

The working FEPs list (Freeze et al. 2010) of the U.S. DOE Used Fuel Disposition (UFD) Campaign is shown in Table B-1. Likely screening decisions are presented for a generic repository in granite.

Table B-1. Likely UFD FEPs Screening Decisions for a Mined Repository in Granite.

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 0.1.02.01 | Timescales of Concern | (unspecified) | Include | |
| 0.1.03.01 | Spatial Domain of Concern | (unspecified) | Include | |
| 0.1.09.01 | Regulatory Requirements and Exclusions | (unspecified) | Include | U.S. regulations will need to be revised |
| 0.1.10.01 | Model Issues | - Conceptual model<br>- Mathematical implementation<br>- Geometry and dimensionality<br>- Process coupling<br>- Boundary and initial conditions | Include | |
| 0.1.10.02 | Data Issues | - Parameterization and values<br>- Correlations<br>- Uncertainty | Include | |
| 1.1.01.01 | Open Boreholes | - Site investigation boreholes (open, improperly sealed)<br>- Preclosure and postclosure monitoring boreholes<br>- Enhanced flow pathways from EBS | Exclude | |
| 1.1.02.01 | Chemical Effects from Preclosure Operations (EBS, EDZ, Host Rock) | - Water contaminants (explosives residue, diesel, organics, etc.)<br>- Water chemistry different than host rock (e.g., oxiding)<br>- Undesirable materials left<br>- Accidents and unplanned events | Exclude | |
| 1.1.02.02 | Mechanical Effects from Preclosure Operations (EBS, EDZ, Host Rock) | - Creation of excavation-disturbed zone (EDZ)<br>- Stress relief<br>- Boring and blasting effects<br>- Rock reinforcement effects (drillholes)<br>- Accidents and unplanned events<br>- Enhanced flow pathways | Exclude | |
| 1.1.02.03 | Thermal-Hydrologic Effects from Preclosure Operations (EBS, EDZ, Host Rock) | - Site flooding<br>- Preclosure ventilation<br>- Accidents and unplanned events | Exclude | |
| 1.1.08.01 | Deviations from Design and Inadequate Quality Control | - Error in waste emplacement (waste forms, waste packages, waste package support materials)<br>- Error in EBS component emplacement<br>- Inadequate excavation / construction (planning, schedule, implementation)<br>- Aborted / incomplete closure of repository<br>- Material and/or component defects | Exclude | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 1.1.10.01 | Control of Repository Site | - Active controls (controlled area)<br>- Retention of records<br>- Passive controls (markers) | Exclude | |
| 1.1.13.01 | Retrievability | (unspecified) | Include | However, the U.S. may reconsider current policy |
| 1.2.01.01 | Tectonic Activity – Large Scale | - Uplift<br>- Folding | Exclude | |
| 1.2.02.01 | Subsidence | (unspecified) | Exclude | |
| 1.2.03.01 | Seismic Activity Impacts EBS and/or EBS Components | - Mechanical damage to EBS (from ground motion, rockfall, drift collapse, fault displacement) | Exclude | |
| 1.2.03.02 | Seismic Activity Impacts Geosphere (Host Rock, Other Geologic Units) | - Altered flow pathways and properties<br>- Altered stress regimes (faults, fractures) | Exclude | |
| 1.2.03.03 | Seismic Activity Impacts Biosphere (Surface Environment, Human Behavior) | - Altered surface characteristics<br>- Altered surface transport pathways<br>- Altered recharge | Exclude | |
| 1.2.04.01 | Igneous Activity Impacts EBS and/or EBS Components | - Mechanical damage to EBS (from igneous intrusion)<br>- Chemical interaction with magmatic volatiles<br>- Transport of radionuclides (in magma, pyroclasts, vents) | Exclude | |
| 1.2.04.02 | Igneous Activity Impacts Geosphere (Host Rock, Other Geologic Units) | - Altered flow pathways and properties<br>- Altered stress regimes (faults, fractures)<br>- Igneous intrusions<br>- Altered thermal and chemical conditions | Exclude | |
| 1.2.04.03 | Igneous Activity Impacts Biosphere (Surface Environment, Human Behavior) | - Altered surface characteristics<br>- Altered surface transport pathways<br>- Altered recharge<br>- Ashfall and ash redistribution | Exclude | |
| 1.2.05.01 | Metamorphism | - Structural changes due to natural heating and/or pressure | Exclude | |
| 1.2.08.01 | Diagenesis | - Mineral alteration due to natural processes | Exclude | |
| 1.2.09.01 | Diapirism | - Plastic flow of rocks under lithostatic loading<br>- Salt / evaporates<br>- Clay | Exclude | |
| 1.2.09.02 | Large-Scale Dissolution | (unspecified) | Exclude | |
| 1.3.01.01 | Climate Change (Natural and Anthropogenic) | - Variations in precipitation and temperature<br>- Long-term global<br>- Short-term regional and local | Include | |
| 1.3.04.01 | Periglacial Effects | - Permafrost<br>- Seasonal freeze/thaw | Include? | Decision would depend upon site latitude |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 1.3.05.01 | Glacial and Ice Sheet Effects | - Glaciation<br>- Isostatic depression<br>- Melt water | Include? | Decision would depend upon site latitude |
| 1.4.01.01 | Human Influences on Climate (Intentional and Accidental) | - Variations in precipitation and temperature<br>- Global, regional, and/or local<br>- Greenhouse gases, ozone layer failure | Exclude | |
| 1.4.02.01 | Human Intrusion (Deliberate and Inadvertent) | - Drilling (resource exploration, …)<br>- Mining / tunneling<br>- Unintrusive site investigation (airborne, surface-based, …) | Include | Need regulatory clarification; 40 CFR 191 and 40 CFR 197 include it but differently |
| 1.4.11.01 | Explosions and Crashes from Human Activities | - War<br>- Sabotage<br>- Testing<br>- Resource exploration / exploitation<br>- Aircraft | Exclude | |
| 1.5.01.01 | Meteorite Impact | - Cratering, host rock removal<br>- Exhumation of waste<br>- Alteration of flow pathways | Exclude | |
| 1.5.01.02 | Extraterrestrial Events | - Solar systems (supernova)<br>- Celestial activity (sun - solar flares, gamma-ray bursters; moon – earth tides)<br>- Alien life forms | Exclude | |
| 1.5.03.01 | Earth Planetary Changes | - Changes in earth's magnetic field<br>- Changes in earth's gravitational field (tides) | Exclude | |
| 2.1.01.01 | Waste Inventory (Radionuclides and Non-Radionuclides) | - Composition<br>- Enrichment / Burn-up | Include | |
| 2.1.01.02 | Radioactive Decay and Ingrowth | (unspecified) | Include | |
| 2.1.01.03 | Heterogeneity of Waste Inventory (Waste Package Scale and Repository Scale) | - Composition<br>- Enrichment / Burn-up<br>- Damaged Area | Include | |
| 2.1.01.04 | Interactions between Co-Located Waste | (unspecified) | Include | |
| 2.1.02.01 | SNF (Commercial, DOE) Degradation (Alteration/Phase Separation, Dissolution/Leaching, Radionuclide Release) | Degradation is dependent on:<br>- Composition<br>- Geometry / structure<br>- Enrichment / burn-up<br>- Surface area<br>- Gap and grain fraction<br>- Damaged area<br>- THC conditions | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.1.02.02 | HLW (Glass, Ceramic, Metal) Degradation (Alteration/Phase Separation, Dissolution/Leaching, Cracking, Radionuclide Release) | Degradation is dependent on: <br> - Composition <br> - Geometry / structure <br> - Surface area <br> - Damaged / cracked area <br> - Mechanical impact <br> - THC conditions | Include | |
| 2.1.02.03 | Degradation of Organic/Cellulosic Materials in Waste | (unspecified) | Exclude | |
| 2.1.02.04 | HLW (Glass, Ceramic, Metal) Recrystallization | (unspecified) | Exclude | |
| 2.1.02.05 | Pyrophoricity or Flammable Gas from SNF or HLW | (unspecified) | Exclude | |
| 2.1.02.06 | SNF Cladding Degradation and Failure | - Initial damage <br> - General corrosion <br> - Microbially influenced corrosion <br> - Localized corrosion <br> - Enhanced corrosion (silica, fluoride) <br> - Stress corrosion cracking <br> - Hydride cracking <br> - Unzipping <br> - Creep <br> - Internal pressure <br> - Mechanical impact | Include | |
| 2.1.03.01 | Early Failure of Waste Packages | - Manufacturing defects <br> - Improper sealing | Include | |
| 2.1.03.02 | General Corrosion of Waste Packages | - Dry-air oxidation <br> - Humid-air corrosion <br> - Aqueous phase corrosion <br> - Passive film formation and stability | Include | |
| 2.1.03.03 | Stress Corrosion Cracking (SCC) of Waste Packages | - Crack initiation, growth and propagation <br> - Stress distribution around cracks | Include | |
| 2.1.03.04 | Localized Corrosion of Waste Packages | - Pitting <br> - Crevice corrosion <br> - Salt deliquescence | Include | |
| 2.1.03.05 | Hydride Cracking of Waste Packages | - Hydrogen diffusion through metal matrix <br> - Crack initiation and growth in metal hydride phases | Exclude? | |
| 2.1.03.06 | Microbially Influenced Corrosion (MIC) of Waste Packages | (unspecified) | Exclude | |
| 2.1.03.07 | Internal Corrosion of Waste Packages Prior to Breach | (unspecified) | Exclude | |
| 2.1.03.08 | Evolution of Flow Pathways in Waste Packages | - Evolution of physical form of waste package <br> - Plugging of cracks in waste packages | Exclude | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.1.04.01 | Evolution and Degradation of Backfill | - Alteration<br>- Thermal expansion / degradation<br>- Swelling / compaction<br>- Erosion / dissolution<br>- Evolution of backfill flow pathways | Include | |
| 2.1.05.01 | Degradation of Seals | - Alteration / degradation / cracking<br>- Erosion / dissolution | Include | |
| 2.1.06.01 | Degradation of Liner / Rock Reinforcement Materials in EBS | - Alteration / degradation / cracking<br>- Corrosion<br>- Erosion / dissolution / spalling | Include | |
| 2.1.07.01 | Rockfall | - Dynamic loading (block size and velocity) | Exclude | |
| 2.1.07.02 | Drift Collapse | - Static loading (rubble volume)<br>- Alteration of seepage<br>- Alteration of EBS flow pathways<br>- Alteration of EBS thermal environment | Exclude | |
| 2.1.07.03 | Mechanical Effects of Backfill | - Protection of other EBS components from rockfall / drift collapse | Exclude | |
| 2.1.07.04 | Mechanical Impact on Backfill | - Rockfall / drift collapse<br>- Hydrostatic pressure<br>- Internal gas pressure | Exclude | |
| 2.1.07.05 | Mechanical Impact on Waste Packages | - Rockfall / drift collapse<br>- Waste package movement<br>- Hydrostatic pressure<br>- Internal gas pressure<br>- Swelling corrosion products | Exclude | The waste package canister is expected to withstand the mechanical pressures in the repository |
| 2.1.07.06 | Mechanical Impact on SNF Waste Form | - Drift collapse<br>- Swelling corrosion products | Exclude | |
| 2.1.07.07 | Mechanical Impact on HLW Waste Form | - Drift collapse<br>- Swelling corrosion products | Exclude | |
| 2.1.07.08 | Mechanical Impact on Other EBS Components (Seals, Liner / Rock Reinforcement Materials, Waste Package Support Materials) | - Rockfall / drift collapse<br>- Movement<br>- Hydrostatic pressure<br>- Swelling corrosion products | Exclude | |
| 2.1.07.09 | Mechanical Effects at EBS Component Interfaces | - Component-to-component contact (static or dynamic) | Exclude | |
| 2.1.07.10 | Mechanical Degradation of EBS | - Floor buckling<br>- Fault displacement<br>- Initial damage from excavation / construction<br>- Consolidation of EBS components<br>- Degradation of waste package support structure<br>- Alteration of EBS flow pathways | Exclude | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.1.08.01 | Flow through the EBS | - Saturated/unsaturated flow<br>- Preferential flow pathways<br>- Density effects on flow<br>- Initial hydrologic conditions<br>- Flow pathways out of EBS | Include | |
| 2.1.08.02 | Flow in and through Waste Packages | - Saturated/unsaturated flow<br>- Movement as thin films or droplets | Include | |
| 2.1.08.03 | Flow in Backfill | - Fracture/matrix flow | Include | |
| 2.1.08.04 | Flow through Seals | (unspecified) | Include | |
| 2.1.08.05 | Flow through Liner / Rock Reinforcement Materials in EBS | (unspecified) | Include | |
| 2.1.08.06 | Alteration and Evolution of EBS Flow Pathways | - Drift collapse<br>- Degradation/consolidation of EBS components<br>- Plugging of flow pathways<br>- Formation of corrosion products<br>- Water ponding | Include | |
| 2.1.08.07 | Condensation Forms in Repository (on Tunnel Roof/Walls, on EBS Components) | - Drift collapse<br>- Degradation/consolidation of EBS components | Exclude | |
| 2.1.08.08 | Capillary Effects in EBS | - Wicking | Include | Resaturation of excavation |
| 2.1.08.09 | Influx/Seepage into the EBS | - Water influx rate (spatial and temporal distribution) | Include | |
| 2.1.09.01 | Chemistry of Water Flowing into the Repository | - Chemical characteristics of influent water (spatial and temporal distribution) | Include | |
| 2.1.09.02 | Chemical Characteristics of Water in Waste Packages | - Water composition (radionuclides, dissolved species, …)<br>- Initial void chemistry (air / gas)<br>- Water chemistry (pH, ionic strength, $pCO_2$, .. )<br>- Reduction-oxidation potential<br>- Reaction kinetics<br>- Influent chemistry (from tunnels and/or backfill)<br>- Evolution of water chemistry / interaction with waste packages | Include | |
| 2.1.09.03 | Chemical Characteristics of Water in Backfill | - Water composition (radionuclides, dissolved species, …)<br>- Water chemistry (pH, ionic strength, $pCO_2$, ..)<br>- Reduction-oxidation potential<br>- Reaction kinetics<br>- Influent chemistry (from tunnels and/or waste package)<br>- Evolution of water chemistry / interaction with backfill | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.1.09.04 | Chemical Characteristics of Water in Tunnels | - Water composition (radionuclides, dissolved species, …)<br>- Water chemistry (pH, ionic strength, pCO2, ..)<br>- Reduction-oxidation potential<br>- Reaction kinetics<br>- Influent chemistry (from near-field host rock)<br>- Initial chemistry (from construction / emplacement)<br>- Evolution of water chemistry / interaction with seals, liner/rock reinforcement materials, waste package support materials<br>- Reduction-oxidation potential | Include | |
| 2.1.09.05 | Chemical Interaction of Water with Corrosion Products (in Waste Packages, in Backfill, in Tunnels) | - Corrosion product formation and composition (waste form, waste package internals, waste package)<br>- Evolution of water chemistry in waste packages, in backfill, and in tunnels | Include | |
| 2.1.09.06 | Chemical Interaction of Water with Backfill (on Waste Packages, in Backfill, in Tunnels) | - Backfill composition and evolution (bentonite, crushed rock, ...)<br>- Evolution of water chemistry in backfill, and in tunnels<br><br>- Enhanced degradation of waste packages (crevice formation) | Include | |
| 2.1.09.07 | Chemical Interaction of Water with Liner / Rock Reinforcement and Cementitious Materials in EBS (in Backfill, in Tunnels) | - Liner composition and evolution (concrete, metal, ...)<br>- Rock reinforcement material composition and evolution (grout, rock bolts, mesh, ...)<br>- Other cementitious materials composition and evolution<br>- Evolution of water chemistry in backfill, and in tunnels | Include | |
| 2.1.09.08 | Chemical Interaction of Water with Other EBS Components (in Waste Packages, in Tunnels) | - Seals composition and evolution<br>- Waste package support composition and evolution (concrete, metal, …)<br>- Other EBS components (other metals (copper), …)<br>- Evolution of water chemistry in backfill, and in tunnels | Include? | |
| 2.1.09.09 | Chemical Effects at EBS Component Interfaces | - Component-to-component contact (chemical reactions)<br>- Consolidation of EBS components | Exclude | |
| 2.1.09.10 | Chemical Effects of Waste-Rock Contact | - Waste-to-host rock contact (chemical reactions)<br>- Component-to-host rock contact (chemical reactions) | Exclude | |
| 2.1.09.11 | Electrochemical Effects in EBS | - Enhanced metal corrosion | Exclude | |
| 2.1.09.12 | Chemical Effects of Drift Collapse | - Evolution of water chemistry in backfill and in tunnels (from altered seepage, from altered thermal-hydrology) | Exclude | |
| 2.1.09.13 | Radionuclide Speciation and Solubility in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Dissolved concentration limits<br>- Limited dissolution due to inclusion in secondary phase<br>- Enhanced dissolution due to alpha recoil | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.1.09.51 | Advection of Dissolved Radionuclides in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Flow pathways and velocity<br>- Advective properties (porosity, tortuosity)<br>- Dispersion<br>- Saturation | Include | |
| 2.1.09.52 | Diffusion of Dissolved Radionuclides in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Gradients (concentration, chemical potential)<br>- Diffusive properties (diffusion coefficients)<br>- Flow pathways and velocity<br>- Saturation | Include | |
| 2.1.09.53 | Sorption of Dissolved Radionuclides in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Surface complexation properties | Include | |
| 2.1.09.54 | Complexation in EBS | - Formation of organic complexants (humates, fulvates, organic waste)<br>- Enhanced transport of radionuclides associated with organic complexants | Include? | |
| 2.1.09.55 | Formation of Colloids in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Formation of intrinsic colloids<br>- Formation of pseudo colloids (host rock fragments, waste form fragments, corrosion products, microbes)<br>- Formation of co-precipitated colloids<br>- Sorption/attachment of radionuclides to colloids (clay, silica, waste form, FeOx, microbes) | Exclude | Salinity and buffer will limit colloidal stability and mobility in EBS. However, deep penetration of glacial melt waters is a scenario that could stabilize colloids and erode buffer. |
| 2.1.09.56 | Stability of Colloids in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Aqueous stability (dependent on water chemistry)<br>- Mechanical stability of colloid (dependent on colloid size, gravitational settling) | Exclude | |
| 2.1.09.57 | Advection of Colloids in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Flow pathways and velocity<br>- Advective properties (porosity, tortuosity)<br>- Dispersion<br>- Saturation<br>- Colloid concentration | Exclude | |
| 2.1.09.58 | Diffusion of Colloids in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Gradients (concentration, chemical potential)<br>- Diffusive properties (diffusion coefficients)<br>- Flow pathways and velocity<br>- Saturation<br>- Colloid concentration | Exclude | |
| 2.1.09.59 | Sorption of Colloids in EBS (in Waste Form, in Waste Package, in Backfill, in Tunnel) | - Sorption of radionuclides to colloids<br>- Surface complexation properties<br>- Sorption of colloids to immobile surfaces | Exclude | |
| 2.1.09.60 | Sorption of Colloids at Air-Water Interface in EBS | (unspecified) | Exclude | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.1.09.61 | Filtration of Colloids in EBS | - Physical filtration (dependent on flow pathways, colloid size) | Exclude | |
| 2.1.09.62 | Radionuclide Transport through Liners and Seals | - Advection<br>- Dispersion<br>- Diffusion<br>- Sorption | Include | |
| 2.1.09.63 | Radionuclide Release from the EBS (Dissolved, Colloidal, and Gas Phase) | - Spatial and temporal distribution of releases to the host rock (due to varying flow pathways and velocities, varying component degradation rates, varying transport properties) | Include | |
| 2.1.10.01 | Microbial Activity in EBS (Natural and Anthropogenic) | - Effects on corrosion<br>- Formation of complexants<br>- Formation of microbial colloids<br>- Formation of biofilms<br>- Biodegradation<br>- Biomass production<br>- Bioaccumulation | Exclude | |
| 2.1.11.01 | Heat Generation in EBS | - Heat transfer (spatial and temporal distribution of temperature and relative humidity) | Include | |
| 2.1.11.02 | Exothermic Reactions in EBS | - Oxidation of SNF<br>- Hydration of concrete | Exclude | |
| 2.1.11.03 | Effects of Backfill on EBS Thermal Environment | - Thermal blanket<br>- Condensation | Include | |
| 2.1.11.04 | Effects of Drift Collapse on EBS Thermal Environment | - Thermal blanket<br>- Condensation | Exclude | |
| 2.1.11.05 | Effects of Influx (Seepage) on Thermal Environment | - Temperature and relative humidity (spatial and temporal distribution) | Exclude | |
| 2.1.11.06 | Thermal-Mechanical Effects on Waste Form and In-Package EBS Components | - Alteration<br>- Cracking<br>- Thermal expansion / stress | Exclude | |
| 2.1.11.07 | Thermal-Mechanical Effects on Waste Packages | - Thermal sensitization / phase changes<br>- Cracking<br>- Thermal expansion / stress / creep | Exclude | |
| 2.1.11.08 | Thermal-Mechanical Effects on Backfill | - Alteration<br>- Cracking<br>- Thermal expansion / stress | Exclude | |
| 2.1.11.09 | Thermal-Mechanical Effects on Other EBS Components (Seals, Liner / Rock Reinforcement Materials, Waste Package Support Structure) | - Alteration<br>- Cracking<br>- Thermal expansion / stress | Exclude | |
| 2.1.11.10 | Thermal Effects on Flow in EBS | - Altered influx/seepage<br>- Altered saturation / relative humidity (dry-out, resaturation)<br>- Condensation | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.1.11.11 | Thermally-Driven Flow (Convection) in EBS | - Convection | Include | |
| 2.1.11.12 | Thermally-Driven Buoyant Flow / Heat Pipes in EBS | - Vapor flow | Include | |
| 2.1.11.13 | Thermal Effects on Chemistry and Microbial Activity in EBS | (unspecified) | Include | |
| 2.1.11.14 | Thermal Effects on Transport in EBS | - Thermal diffusion (Soret effect)<br>- Thermal osmosis | Exclude | |
| 2.1.12.01 | Gas Generation in EBS | - Repository pressurization<br>- Mechanical damage to EBS components<br>- He generation from waste from alpha decay<br>- H2 generation from waste package corrosion<br>- CO2, CH4, and H2S generation from microbial degradation | Include | |
| 2.1.12.02 | Effects of Gas on Flow through the EBS | - Two-phase flow<br>- Gas bubbles | Include | |
| 2.1.12.03 | Gas Transport in EBS | - Gas phase transport<br>- Gas phase release from EBS | Include | |
| 2.1.12.04 | Gas Explosions in EBS | (unspecified) | Exclude | |
| 2.1.13.01 | Radiolysis (in Waste Package, in Backfill, and in Tunnel) | - Gas generation<br>- Altered water chemistry | Exclude | |
| 2.1.13.02 | Radiation Damage to EBS Components (in Waste Form, in Waste Package, in Backfill, in Other EBS Components) | - Enhanced waste form degradation<br>- Enhanced waste package degradation<br>- Enhanced backfill degradation<br>- Enhanced degradation of other EBS components (liner/rock reinforcement materials, seals, waste support structure) | Exclude | |
| 2.1.13.03 | Radiological Mutation of Microbes | (unspecified) | Exclude | |
| 2.1.14.01 | Criticality In-Package | - Formation of critical configuration | Exclude | |
| 2.1.14.02 | Criticality in EBS or Near-Field | - Formation of critical configuration | Exclude | |
| 2.2.01.01 | Evolution of EDZ | - Lateral extent, heterogeneities<br>- Physical properties<br>- Flow pathways<br>- Chemical characteristics of groundwater in EDZ<br>- Radionuclide speciation and solubility in EDZ<br>- Thermal-mechanical effects<br>- Thermal-chemical alteration | Include | |
| 2.2.02.01 | Stratigraphy and Properties of Host Rock | - Rock units<br>- Thickness, lateral extent, heterogeneities, discontinuities, contacts<br>- Physical properties<br>- Flow pathways | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.2.03.01 | Stratigraphy and Properties of Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Rock units<br>- Thickness, lateral extent, heterogeneities, discontinuities, contacts<br>- Physical properties<br>- Flow pathways | Include | |
| 2.2.05.01 | Fractures (Host Rock, and Other Geologic Units) | - Rock properties | Include | |
| 2.2.05.02 | Faults (Host Rock, and Other Geologic Units) | - Rock properties | Include | |
| 2.2.05.03 | Alteration and Evolution of NBS Flow Pathways (Host Rock and Other Geologic Units) | - Changes in rock properties<br>- Changes in faults<br>- Changes in fractures<br>- Plugging of flow pathways<br>- Changes in saturation | Exclude | |
| 2.2.07.01 | Mechanical Effects on Host Rock | - From subsidence<br>- From salt creep<br>- From clay deformation<br>- From granite deformation (rockfall / drift collapse into tunnels)<br>- Chemical precipitation / dissolution | Exclude | |
| 2.2.07.02 | Mechanical Effects on Other Geologic Units | - From subsidence<br>- Chemical precipitation / dissolution | Exclude | |
| 2.2.08.01 | Flow through the Host Rock | - Saturated flow<br>- Fracture flow / matrix imbibition<br>- Unsaturated flow (fingering, capillarity, episodicity, perched water)<br>- Preferential flow pathways<br>- Density effects on flow<br>- Flow pathways out of host rock | Include | |
| 2.2.08.02 | Flow through the Other Geologic Units (Confining Units and Aquifers) | - Saturated flow<br>- Fracture flow / matrix imbibition<br>- Unsaturated flow (fingering, capillarity, episodicity, perched water)<br>- Preferential flow pathways<br>- Flow pathways out of other geologic units | Include | |
| 2.2.08.03 | Effects of Recharge on NBS Flow (Host Rock and Other Geologic Units) | - Infiltration rate<br>- Water table rise/decline | Include | |
| 2.2.08.04 | Effects of Repository Excavation on Flow through the Host Rock | - Saturated flow (flow sink)<br>- Unsaturated flow (capillary diversion, drift shadow)<br>- Influx/Seepage into EBS (film flow, enhanced seepage) | Include | |
| 2.2.08.05 | Condensation Forms in Host Rock | - Condensation cap<br>- Shedding | Exclude | |
| 2.2.08.06 | Flow through EDZ | - Saturated / unsaturated flow<br>- Fracture / matrix flow | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.2.08.07 | Mineralogic Dehydration | - Dehydration reactions release water and may lead to volume changes | Exclude | |
| 2.2.08.08 | Groundwater Discharge to Biosphere Boundary | - Surface discharge (water table, capillary rise, surface water)<br>- Flow across regulatory boundary | Include | |
| 2.2.08.09 | Groundwater Discharge to Well | - Human use (drinking water, bathing water, industrial)<br>- Agricultural use (irrigation, animal watering) | Include | |
| 2.2.09.01 | Chemical Characteristics of Groundwater in Host Rock | - Water composition (radionuclides, dissolved species, …)<br>- Water chemistry (temperature, pH, Eh, ionic strength …)<br>- Reduction-oxidation potential<br>- Reaction kinetics<br>- Interaction with EBS<br>- Interaction with host rock | Include | |
| 2.2.09.02 | Chemical Characteristics of Groundwater in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Water composition (radionuclides, dissolved species, …)<br>- Water chemistry (temperature, pH, Eh, ionic strength …)<br>- Reduction-oxidation potential<br>- Reaction kinetics<br>- Interaction with other geologic units | Include | |
| 2.2.09.03 | Chemical Interactions and Evolution of Groundwater in Host Rock | - Host rock composition and evolution (granite, clay, salt …)<br>- Evolution of water chemistry in host rock<br>- Chemical effects on density<br>- Interaction with EBS<br>- Reaction kinetics<br>- Mineral dissolution/precipitation<br>- Redissolution of precipitates after dry-out | Include | |
| 2.2.09.04 | Chemical Interactions and Evolution of Groundwater in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Host rock composition and evolution (granite, clay, salt …)<br>- Evolution of water chemistry in host rock<br>- Chemical effects on density<br>- Reaction kinetics<br>- Mineral dissolution/precipitation<br>- Recharge chemistry | Include | |
| 2.2.09.05 | Radionuclide Speciation and Solubility in Host Rock | - Dissolved concentration limits | Include | |
| 2.2.09.06 | Radionuclide Speciation and Solubility in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Dissolved concentration limits | Include | |
| 2.2.09.51 | Advection of Dissolved Radionuclides in Host Rock | - Flow pathways and velocity<br>- Advective properties (porosity, tortuosity)<br>- Dispersion<br>- Matrix diffusion<br>- Saturation | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.2.09.52 | Advection of Dissolved Radionuclides in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Flow pathways and velocity<br>- Advective properties (porosity, tortuosity)<br>- Dispersion<br>- Matrix diffusion<br>- Saturation | Include | |
| 2.2.09.53 | Diffusion of Dissolved Radionuclides in Host Rock | - Gradients (concentration, chemical potential)<br>- Diffusive properties (diffusion coefficients)<br>- Flow pathways and velocity<br>- Saturation | Include | |
| 2.2.09.54 | Diffusion of Dissolved Radionuclides in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Gradients (concentration, chemical potential)<br>- Diffusive properties (diffusion coefficients)<br>- Flow pathways and velocity<br>- Saturation | Include | |
| 2.2.09.55 | Sorption of Dissolved Radionuclides in Host Rock | - Surface complexation properties<br>- Flow pathways and velocity<br>- Saturation | Include | |
| 2.2.09.56 | Sorption of Dissolved Radionuclides in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Surface complexation properties<br>- Flow pathways and velocity<br>- Saturation | Include | |
| 2.2.09.57 | Complexation in Host Rock | - Presence of organic complexants (humates, fulvates, carbonates, …)<br>- Enhanced transport of radionuclides associated with organic complexants | Include | |
| 2.2.09.58 | Complexation in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Presence of organic complexants (humates, fulvates, carbonates, …)<br>- Enhanced transport of radionuclides associated with organic complexants | Include | |
| 2.2.09.59 | Colloidal Transport in Host Rock | - Flow pathways and velocity<br>- Saturation<br>- Advection<br>- Dispersion<br>- Diffusion<br>- Sorption<br>- Colloid concentration | Include | Although salinity will likely limit colloid stability, deep penetration of glacial melt waters could stabilize colloids. |
| 2.2.09.60 | Colloidal Transport in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Flow pathways and velocity<br>- Saturation<br>- Advection<br>- Dispersion<br>- Diffusion<br>- Sorption<br>- Colloid concentration | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.2.09.61 | Radionuclide Transport through EDZ | - Advection<br>- Dispersion<br>- Diffusion<br>- Sorption | Include | |
| 2.2.09.62 | Dilution of Radionuclides in Groundwater (Host Rock and Other Geologic Units) | - Mixing with uncontaminated groundwater<br>- Mixing at withdrawal well | Include | |
| 2.2.09.63 | Dilution of Radionuclides with Stable Isotopes (Host Rock and Other Geologic Units) | - Mixing with stable and/or naturally occurring isotopes of the same element | Include | Could be important for radioisotopes such as I-129 |
| 2.2.09.64 | Radionuclide Release from Host Rock (Dissolved, Colloidal, and Gas Phase) | - Spatial and temporal distribution of releases to the other geologic units (due to varying flow pathways and velocities, varying transport properties) | Include | |
| 2.2.09.65 | Radionuclide Release from Other Geologic Units (Dissolved, Colloidal, Gas Phase) | - Spatial and temporal distribution of releases to the biosphere (due to varying flow pathways and velocities, varying transport properties) | Include | |
| 2.2.10.01 | Microbial Activity in Host Rock | - Formation of complexants<br>- Formation and stability of microbial colloids<br>- Biodegradation<br>- Bioaccumulation | Exclude | |
| 2.2.10.02 | Microbial Activity in Other Geologic Units (Non-Host-Rock) (Confining Units and Aquifers) | - Formation of complexants<br>- Formation and stability of microbial colloids<br>- Biodegradation<br>- Bioaccumulation | Exclude | |
| 2.2.11.01 | Thermal Effects on Flow in NBS (Repository-Induced and Natural Geothermal) | - Altered saturation / relative humidity (dry-out, resaturation)<br>- Altered gradients, density, and/or flow pathways<br>- Vapor flow<br>- Condensation | Include | |
| 2.2.11.02 | Thermally-Driven Flow (Convection) in NBS | - Convection | Include | |
| 2.2.11.03 | Thermally-Driven Buoyant Flow / Heat Pipes in NBS | - Vapor flow | Include | |
| 2.2.11.04 | Thermal Effects on Chemistry and Microbial Activity in NBS | - Mineral precipitation / dissolution<br>- Altered solubility | Include | |
| 2.2.11.05 | Thermal Effects on Transport in NBS | - Thermal diffusion (Soret effect)<br>- Thermal osmosis | Exclude | |
| 2.2.11.06 | Thermal-Mechanical Effects on NBS | - Thermal expansion / compression<br>- Altered properties of fractures, faults, rock matrix | Exclude | |
| 2.2.11.07 | Thermal-Chemical Alteration of NBS | - Mineral precipitation / dissolution<br>- Altered properties of fractures, faults, rock matrix<br>- Alteration of minerals / volume changes<br>- Formation of near-field chemically altered zone (rind) | Include | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.2.12.01 | Gas Generation in NBS | - Degassing (clathrates, deep gases)<br>- Microbial degradation of organics | Exclude | |
| 2.2.12.02 | Effects of Gas on Flow through the NBS | - Altered gradients and/or flow pathways<br>- Vapor/air flow<br>- Two-phase flow<br>- Gas bubbles | Include | |
| 2.2.12.03 | Gas Transport in NBS | - Gas phase transport<br>- Gas phase release from geosphere | Include | |
| 2.2.14.01 | Criticality in Far-Field | - Formation of critical configuration | Exclude | |
| 2.3.01.01 | Topography and Surface Morphology | - Recharge and discharge areas | Include | |
| 2.3.02.01 | Surficial Soil Type | - Physical and chemical attributes | Include | |
| 2.3.04.01 | Surface Water | - Lakes, rivers, springs<br>- Dams, reservoirs, canals, pipelines<br>- Coastal and marine features<br>- Water management activities | Exclude | |
| 2.3.05.01 | Biosphere Characteristics | - Climate<br>- Soils<br>- Flora and fauna<br>- Microbes<br>- Evolution of biosphere (natural, anthropogenic – e.g., acid rain) | Include | |
| 2.3.07.01 | Erosion | - Weathering<br>- Denudation<br>- Subsidence | Exclude | |
| 2.3.07.02 | Deposition | - Weathering | Exclude | |
| 2.3.07.03 | Animal Intrusion into Repository | (unspecified) | Exclude | |
| 2.3.08.01 | Precipitation | - Spatial and temporal distribution | Include | |
| 2.3.08.02 | Surface Runoff and Evapotranspiration | - Runoff, impoundments, flooding, increased recharge<br>- Evaporation<br>- Transpiration (root uptake) | Include | |
| 2.3.08.03 | Infiltration and Recharge | - Spatial and temporal distribution<br>- Effect on hydraulic gradient<br>- Effect on water table elevation | Include | |
| 2.3.09.01 | Chemical Characteristics of Soil and Surface Water | - Altered recharge chemistry (natural)<br>- Altered recharge chemistry (anthropogenic – e.g., acid rain) | Exclude | |
| 2.3.09.02 | Radionuclide Speciation and Solubility in Biosphere | - Dissolved concentration limits | Exclude | |
| 2.3.09.03 | Radionuclide Alteration in Biosphere | - Altered physical and chemical properties<br>- Isotopic dilution | Exclude | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 2.3.09.51 | Atmospheric Transport through Biosphere | - Radionuclide transport in air, gas, vapor, particulates, aerosols<br>- Processes include: wind, plowing, irrigation, degassing, saltation, precipitation | Include | |
| 2.3.09.52 | Surface Water Transport through Biosphere | - Radionuclide transport and mixing in surface water<br>- Processes include: lake mixing, river flow, spring discharge, aeration, sedimentation, dilution | Exclude | |
| 2.3.09.53 | Soil and Sediment Transport through Biosphere | - Radionuclide transport in on soil and sediments<br>- Processes include: fluvial (runoff, river flow), eolian (wind), glaciation, bioturbation (animals) | Include | |
| 2.3.09.54 | Radionuclide Accumulation in Soils | - Leaching/evaporation from discharge (well, groundwater upwelling)<br>- Deposition from atmosphere or water (irrigation, runoff) | Include | |
| 2.3.09.55 | Recycling of Accumulated Radionuclides from Soils to Groundwater | (unspecified) | Include | |
| 2.3.10.01 | Microbial Activity in Biosphere | - Effect on biosphere characteristics<br>- Effect on transport through biosphere | Exclude | |
| 2.3.11.01 | Effects of Repository Heat on Biosphere | (unspecified) | Exclude | |
| 2.4.01.01 | Human Characteristics | - Physiology<br>- Metabolism<br>- Adults, children | Include | |
| 2.4.01.02 | Human Evolution | - Changing human characteristics<br>- Sensitization to radiation<br>- Changing lifestyle | Exclude | |
| 2.4.04.01 | Human Lifestyle | - Diet and fluid intake (food, water, tobacco/drugs, etc.)<br><br>- Dwellings<br>- Household activities<br>- Leisure activities | Include | |
| 2.4.08.01 | Land and Water Use | - Agricultural (irrigation, plowing, fertilization, crop storage, greenhouses, hydroponics)<br>- Farms and Fisheries (feed, water, soil)<br>- Urban / industrial (development, energy production, earthworks, population density)<br>- Natural / wild (grasslands, forests, bush, surface water) | Include | |
| 2.4.08.02 | Evolution of Land and Water Use | - New practices (agricultural, farming, fisheries)<br>- Technological developments<br>- Social developments (new/expanded communities) | Exclude | |

| UFD FEP Number | Description | Associated Processes | Likely Screening Decision | Comment |
|---|---|---|---|---|
| 3.3.01.01 | Radionuclides in Biosphere Media | - Soil<br>- Surface water<br>- Air<br>- Plant uptake<br>- Animal (livestock, fish) uptake | Include | |
| 3.3.01.02 | Radionuclides in Food Products | - Diet and fluid sources (location, degree of contamination, dilution with uncontaminated sources)<br>- Foodstuff and fluid processing and preparation (water filtration, cooking techniques) | Include | |
| 3.3.01.03 | Radionuclides in Non-Food Products | - Dwellings (location, building materials and sources, fuel sources)<br>- Household products (clothing and sources, furniture and sources, tobacco, pets)<br>- Biosphere media | Exclude | |
| 3.3.04.01 | Ingestion | - Food products<br>- Soil, surface water | Include | |
| 3.3.04.02 | Inhalation | - Gases and vapors<br>- Suspended particulates (dust, smoke, pollen) | Include | |
| 3.3.04.03 | External Exposure | - Non-food products<br>- Soil, surface water | Include | |
| 3.3.06.01 | Radiation Doses | - Exposure rates (ingestion, inhalation, external exposure)<br>- Dose conversion factors<br>- Gases and vapors<br>- Suspended particulates (dust, smoke, pollen) | Include | |
| 3.3.06.02 | Radiological Toxicity and Effects | - Human health effects from radiation doses | Include | |
| 3.3.06.03 | Non-Radiological Toxicity and Effects | - Human health effects from non-radiological toxicity | Exclude | |

## B.2   References

Freeze, G. A., P. E. Mariner, J. E. Houseworth, and J. C. Cunnane 2010. *Used Fuel Disposition Campaign Features, Events, and Processes (FEPs): FY10 Progress Report*. SAND2010-5902. Sandia National Laboratories, Albuquerque, New Mexico.

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